Commentary on the Management of Fish Habitat in Northern Canada: Information Requirements and Policy Considerations Regarding Diamond, Oil Sands and Placer Mining

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COMMENTARY ON THE MANAGEMENT OF FISH HABITAT IN NORTHERN CANADA: INFORMATION REQUIREMENTS AND POLICY CONSIDERATIONS REGARDING DIAMOND, OIL SANDS AND PLACER MINING

by

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PREFACE

This report has been prepared in response to the need for information that will assist the management of fish and their habitat in northern Canada, and especially that relating to exploration and mining for diamonds. Increased exploration and mining activity is occurring across Canada and potential diamond mines have been identified in Nunavut, Alberta, Saskatchewan, Ontario and Quebec. Diamond mining is currently occurring in the Northwest Territories.

Fisheries and Oceans Canada (DFO) has authorized the elimination or partial destruction of 26 lakes for diamond mining since 1997, and an additional 5 for the metal mining sector. The elimination of lakes is escalating in relation to the needs of the mining sector in Canada.

The application of appropriate compensatory and restorative techniques is a fundamental requirement of developments that impact fish and their habitat. This poses an especially significant and unique challenge in northern Canada because of the prevalent climatic conditions and the paucity of applicable knowledge.

Although the focus of this report is related to diamond mining in the Arctic, attention is also given to oil sands and placer gold mining; other major industrial activities in northern Canada that impact upon fish and their habitat. The elimination and degradation of lakes across Canada for metal mine tailings disposal and access to ore exemplify decisions that have facilitated industrial developments.

One of the main objectives of this report was to identify deficiencies in our knowledge with respect to fish and their habitat in northern Canada, and the implications of habitat alteration and destruction to aquatic systems. Comments are also provided on related habitat compensatory and restorative measures and the consistency of application of the *Fisheries Act* and use of the "Policy for the Management of Fish Habitat" (Department of Fisheries and Oceans 1986).

Fifty-one representatives of the diamond, placer gold, and oil sands mining industries, scientific and operational colleagues in governments and academia, and certain independent people collaborated in the production of this report by providing valuable information and insight.

It is apparent that there is still much to know about the biology of Canada's north, the implications of climate and habitat change to aquatic organisms, and the appropriateness and adequacy of compensation, restoration and development activities. It is hoped that the information within this report will draw attention to these deficiencies in our understanding of the ecology of certain aquatic organisms and thereby help guide scientific research and assist habitat management.

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ABSTRACT

Birtwell, I.K., Samis, S.C., and Khan, N.Y. 2005. Commentary on the management of fish habitat in northern Canada: information requirements and policy considerations regarding diamond, oil sands, and placer mining. Can. Tech. Rep. Fish. Aquat. Sci. 2606: xxiii + 333 p.

This report identifies scientific and management information needs that are necessary in order to make consistent and defensible decisions which conform with the intent of the "Policy for the Management of Fish Habitat" ("Habitat Policy", Department of Fisheries and Oceans 1986), and that assist with the facilitation of responsible mining activities compliant with the *Fisheries Act* (Government of Canada 1985). Increasing mining activities in Canada and the associated need to eliminate and degrade lakes and watercourses provided the stimulus for the assessment. Since the 1950s, 50 lakes, in whole or in part, have been eliminated or approved for elimination. Of these lakes, 62% (31) were in the last decade, and an additional 20 are to be considered for approval within the next 4 years.

Compensation and restoration for the destruction of lake and stream habitats pose significant scientific and management challenges, especially in northern Canada where ecological knowledge and understanding are rudimentary.

Literature on the biology of selected Arctic species of fish and on watershed ecology was assessed (Khan et al. 2005). Assessments of this literature were combined with opinions from diamond, placer gold and oil sands mining representatives, government scientists and habitat practitioners, and other knowledgeable people in private and public sectors (Samis et al. 2005). Together, these reports provide reference information and an audit trail regarding the statements contained in this technical report and a summary document (Birtwell et al. 2005).

The amalgamation of opinions and published information provide the basis for conclusions and recommendations regarding the management of fish habitat, and the scientific information and procedural requirements to effectively fulfill Departmental responsibilities in this regard. Among the numerous conclusions drawn and recommendations made in this review, the following are considered to be of paramount importance:

- **Damage to fish habitat in pristine regions of the Canadian Arctic is occurring and escalating and there is currently no assurance that habitat compensatory and restorative measures will be effective in meeting the Habitat Policy requirements regarding fish habitat productive capacity, which are linked to the *Fisheries Act* in order to sustain fish productivity.**

- **The inadequacy of even basic ecological knowledge, the absence of validations of habitat compensatory and restorative measures regarding habitat linkages to**

fish productivity, and no examples of whole lake restoration and compensation to guide developments forecast irreparable harm.

- **Significantly more knowledge and understanding of the basic biology and habitat requirements of Arctic species are required, especially in view of the increasing human development in northern Canada, global environmental changes and the need to determine and assess their cumulative effects. Provision of this knowledge and understanding will assist decision making, reduce risk and uncertainty and facilitate mitigation, compensatory and restorative measures designed to sustain aquatic resources in this and other remote regions in Canada.**

- **Co-operative and collaborative ventures with industries, governments, academia and Aboriginal communities should provide for the acquisition of new information. Opportunities currently exist for this to occur. Multi-stakeholder consortia should be the mechanisms for fund acquisition and disbursement.**

- **The successful application of the Habitat Policy to developments in regions of Canada where there is a paucity or absence of basic information for making sound decisions is jeopardized in view of the potential uncertainty of success of habitat mitigation, compensatory and restorative measures. In this context consideration must be given to learning from, and adapting to, experimental decision making. This will be accomplished through committed research and monitoring which is appropriately scaled over time and in space to meet stated Departmental objectives.**

- **Compensatory, restorative, and related techniques, measurements and evaluation, are priority research topics that meet with approval from all industry sectors, the scientific community and habitat practitioners.**

RÉSUMÉ

Birtwell, I.K., Samis, S.C., and Khan, N.Y. 2005. Commentary on the management of fish habitat in northern Canada: information requirements and policy considerations regarding diamond, oil sands, and placer mining. Can. Tech. Rep. Fish. Aquat. Sci. 2606: xxiii + 333 p.

Nous établissons les renseignements scientifiques et l'information de gestion requis pour être en mesure de prendre des décisions cohérentes et défendables, conformes à l'esprit de la Politique de gestion de l'habitat du poisson (Ministère des Pêches et des Océans, 1986) et permettant de faciliter des activités minières responsable conformes à la *Loi sur les pêches* (Gouvernement du Canada 1985). Les activités minières à la hausse au Canada et le besoin résultant de détruire et de dégrader des lacs et des cours d'eau sont à l'origine de la présente évaluation. Depuis les années 1950, 50 lacs, en entier ou en partie, ont été détruits ou leur destruction a été approuvée. De ceux-ci, 62% (31) l'ont été au cours de la dernière décennie, et l'approbation de la destruction de 20 autres sera considérée au cours des quatre prochaines années.

La compensation de la destruction et la restauration de parcelles d'habitat lacustres et lotiques posent d'importants défis sur le plan scientifique et gestionnel, en particulier dans le nord du Canada, car les connaissances et la compréhension de l'écologie de cette région sont plutôt rudimentaires.

Les ouvrages scientifiques sur la biologie de certaines espèces de poissons de l'Arctique et l'écologie des bassins versants ont été évalués (Khan et al. 2005). Les résultats de ces évaluations ont été combinés aux opinions de représentants de mines de diamants, d'or placérien et de sable pétrolière, de scientifiques de l'État et d'agents responsables de l'habitat, ainsi que d'autres personnes bien informées des secteurs public et privé (Samis et al. 2005). Ensemble, ces rapports constituent une source de référence et une piste de vérification des énoncés présentés dans le présent rapport technique et un document récapitulatif (Birtwell et al. 2005).

La fusion des opinions et des renseignements publiés a permis d'établir le fondement des conclusions et des recommandations concernant la gestion de l'habitat du poisson, ainsi que les renseignements scientifiques et les modalités d'application pour s'acquitter efficacement des responsabilités du Ministère dans ce sens. Parmi les nombreuses conclusions tirées et les recommandations formulées dans le cadre du présent examen, nous considérons les suivantes comme les plus importantes :

- **Des dommages à l'habitat du poisson dans les régions sauvages de l'Arctique canadien se produisent et s'intensifient, et rien ne permet d'assurer à l'heure actuelle que les mesures de compensation et de restauration de l'habitat seront efficaces pour ce qui est de satisfaire aux exigences de la *Politique de gestion de l'habitat du poisson* découlent de la *Loi sur les pêches* relatives à la productivité soutenue du poisson.**

- **Le manque de connaissances écologiques même les plus fondamentales, le fait que les mesures de compensation et de restauration de l'habitat n'aient pas été validées en regard des liens entre l'habitat et la productivité du poisson et l'absence d'exemples de restauration et de compensation d'un lac entier pour orienter les activités de développement laissent prévoir des dommages irréparables.**

- **Des connaissances et une compréhension nettement plus poussées de la biologie fondamentale et des besoins au plan de l'habitat des espèces de l'Arctique sont requises, en particulier à la lumière des activités de développement à la hausse dans le nord du Canada, le changement de l'environnement planétaire et le besoin de déterminer et d'évaluer leurs effets cumulatifs. Ces connaissances et cette compréhension permettront de prendre des décisions, de réduire les risques et les incertitudes et de faciliter la mise en oeuvre de mesures d'atténuation, de compensation et de restauration visant à assurer la subsistance des ressources aquatiques dans cette région et dans d'autres régions éloignées du Canada.**

- **Des projets de collaboration réunissant des intervenants d'industries, de gouvernements, d'universités et de collectivités autochtones devraient permettre d'acquérir cette nouvelle information. Il existe actuellement des opportunités de le faire. Les fonds nécessaires devraient provenir de consortiums.**

- **L'application fructueuse de la *Politique de gestion de l'habitat du poisson* aux activités de développement dans les régions du Canada pour lesquelles on ne dispose que peu ou pas de données de base pour prendre des décisions judicieuses est compromise en raison de l'incertitude du succès des mesures d'atténuation, de compensation et de restauration. Dans ce contexte, il faut considérer que l'apprentissage se fera par le biais de la prise de décision expérimentale et qu'il faudra s'y adapter. Cela se fera pas le truchement de recherches et d'une surveillance ciblées, échelonnées adéquatement dans l'espace et le temps, de sorte à satisfaire aux objectifs énoncés par le Ministère.**

- **Les techniques de compensation et de restauration et les techniques connexes, les mesures et l'évaluation constituent des sujets de recherche prioritaires qui reçoivent l'agrément des secteurs de l'industrie, du monde scientifique et des spécialistes de l'habitat.**

INTRODUCTION

The initiation of diamond mining in Canada's north prompted concern over the management of fish habitat during the exploration, development and mining phases. Fish habitat has been, and will be, adversely affected through *Fisheries Act* authorizations that enable mining development. These effects range from those that are subtle, to the elimination of lakes and watercourses. Accordingly, and depending upon the extent and duration of these changes, there will be implications to the well being of aquatic resources that utilize these habitats, and the success of compensatory measures.

The challenge for those who must manage the habitat of fish is to understand the effects of changes in habitat, or its elimination, in relation to the protection and perpetuation of fish. Implicit in this approach is the requirement to provide a healthy environment that in turn produces healthy individuals thereby sustaining populations, stocks and species in the rigorous climatic conditions of Canada's north. Thus knowledge is required, not only to assist decisions that address the initial impacts of mining activities, but also to understand and utilize mitigation and compensatory and restorative measures to maintain the productivity of fish habitat.

Under the "Policy for the Management of Fish Habitat" ("Habitat Policy," Department of Fisheries and Oceans 1986), that was developed under the auspices of the *Fisheries Act* (Government of Canada 1985), Fisheries and Oceans Canada's (DFO's) long-term policy objective is the achievement of an overall net gain of the productive capacity of fish habitats. Progress toward this objective can be achieved through the active conservation of the current productive capacity of habitats, the restoration of damaged fish habitats and the development of habitats.

Fish habitat conservation is the first goal of the Habitat Policy (refer to Figure 1). It is to be achieved by maintaining the current productive capacity of fish habitats supporting Canada's fisheries resources such that fish suitable for human consumption may be produced. The no net loss principle is fundamental to the habitat conservation goal. Under this principle, the Department will strive to balance unavoidable habitat losses with habitat replacement on a project-by-project basis so that further reductions to Canada's fisheries resources due to habitat loss or damage may be prevented. Under the Habitat Policy's second goal, fish habitat restoration, rehabilitation of the productive capacity of fish habitats will be conducted in selected areas where economic or social benefits can be achieved through the fisheries resource. Under the third goal, fish habitat development, and improvement and creation of habitat in selected areas will occur where the production of fisheries resources can be increased for the social or economic benefit of Canadians.

The Department is guided by the following hierarchy of preferences to achieve no net loss (NNL) of the productive capacity of fish habitat. For the application of this principle, the first preference of the Department will be to maintain, without disruption, the natural productive capacity of the habitat(s) in question by avoiding any loss or harmful alteration at the site of the proposed project or activity. It may be achieved by

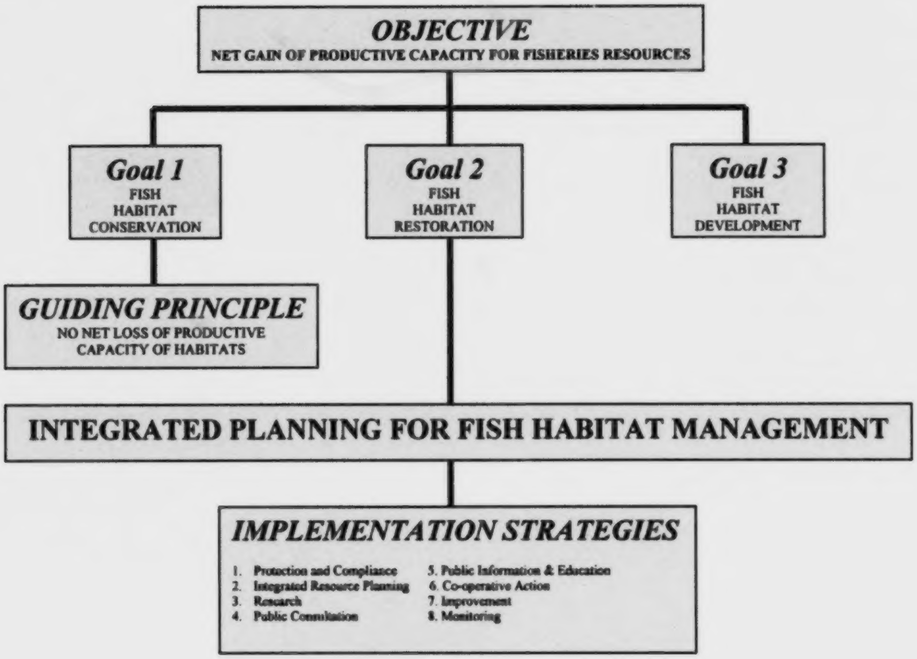

OBJECTIVE
NET GAIN OF PRODUCTIVE CAPACITY FOR FISHERIES RESOURCES
Goal 1
FISH HABITAT CONSERVATION
Goal 2
FISH HABITAT RESTORATION
Goal 3
FISH HABITAT DEVELOPMENT
GUIDING PRINCIPLE
NO NET LOSS OF PRODUCTIVE CAPACITY OF HABITATS
INTEGRATED PLANNING FOR FISH HABITAT MANAGEMENT
IMPLEMENTATION STRATEGIES
1. Protection and Compliance
2. Integrated Resource Planning
3. Research
4. Public Consultation
5. Public Information & Education
6. Co-operative Action
7. Improvement
8. Monitoring

Figure 1. Diagram of the Habitat Policy framework for fish habitat management, its objectives, goals, and strategies (adapted from Department of Fisheries and Oceans 1986).

encouraging the proponent to redesign or relocate the project. Only after it is proven impossible or impractical to maintain the same level of habitat productive capacity using the approaches outlined above would the Department accede to the exploration of compensatory options (Fisheries and Oceans Canada 2004*a*). Firstly, the possibilities for like-for-like habitat compensation should be assessed; that is, replacing natural habitat at or near the site of habitat impact. Should this not be feasible, then secondly, consideration may be given to creating or increasing the productive capacity of unlike habitat in the same ecological unit. Thirdly, consideration may be given to placing the "replaced habitat" at a location away from the impacted area, or increasing the productivity of existing habitat for the affected fisheries resources, if reliable techniques are available. In those rare cases where it is not technically feasible to avoid potential damage to habitats, or to compensate for the habitat itself, the Department, under certain conditions, would consider proposals to compensate in the form of artificial production to supplement the fishery resource. It is apparent, however, that there are inherent risks in the achievement of the Habitat Policy objective, which is a net gain of productive capacity for fisheries resources (Figure 1). This is due, in part, to the paucity of relevant information in those northern regions of Canada where mining occurs.

The objectives of this report were to identify scientific and management information needs so that consistent and defensible decisions may be made that conform with the intent of the Habitat Policy (Department of Fisheries and Oceans 1986), and that assist with the facilitation of responsible mining activities compliant with the *Fisheries Act* (Government of Canada 1985). Accordingly, comments are provided on placer gold, oil sands, and diamond mining, which are significant industrial activities that affect fish habitat in northern Canada, and pose similar questions and challenges. However, unlike the diamond mining industry, placer gold and oil sands mining have been in existence for many decades. The destruction of lakes is a common requirement to the diamond and metal mining sectors in Canada. Descriptions of lakes that have been eliminated, and comments on regulatory issues and associated compensation are provided in relation to these industries.

The "Speech from the Throne" in October 2004 (Government of Canada 2004*a*) emphasized a comprehensive strategy for the north. DFO will have a key role in the facilitation of economic development through observations of climate change and impacts on northern aquatic ecosystems, and co-operation with the international circumpolar community to promote "northern science and technology and Arctic development in a global context." The recommendations presented in this report are supportive of statements contained within that speech.

APPROACHES TO TASK

To meet the objectives of the project, this report gathers and assesses information into sections dealing with the biology and ecology of selected species of fish, watershed ecology, cumulative effects, historical and current details on mining industries, their regulation and impact, habitat science, management, and policies (including aspects of mitigation, compensation, and restoration). A primary focus of the study was to identify critical gaps in knowledge relating to the biology and ecology of fish species in waters in the Canadian Arctic, and, specifically, to relate them to the management of fish habitat. As noted by Power (1997), "the geographical boundaries of the Arctic are not always explicitly defined" in the literature. Most dictionaries define the Arctic as the "regions around the North Pole" or area north of the Arctic Circle. For the purposes of this report the Arctic in Canada was viewed as the region north of 60°N – the provincial-territorial northern boundary (Reist 1997). As such, this region includes the tundra (Barrenlands) and a substantive portion of the sub-Arctic biome consisting of boreal forest (Reist 1997).

Decisions have been made to permit changes to the productivity of Arctic waters to accommodate industry, and concerns exist over the potential short- and long-term impacts at the local and watershed level. Furthermore, the projected expansion of the diamond mining industry and other industries in northern Canada reveals the potential for more site-specific and cumulative impacts on aquatic productivity. While there are obvious incentives for the progression of industrial development, there is nevertheless a requirement that these activities occur in a responsible manner. Such initiatives would be within the authority of existing legislation, and with due regard and sensitivity for affected communities.

REQUIREMENT FOR KNOWLEDGE AND UNDERSTANDING

This report examines and reveals deficiencies in our knowledge of northern aquatic ecology within the context of industrial development, and habitat management requirements under the *Fisheries Act*. It is based on information obtained from the scientific literature, and the comments of people who are knowledgeable about certain industrial operations and their potential impact on fish and their habitat in northern

Canada including comments from "habitat practitioners" within DFO. The report includes, therefore, a blend of quantitative information and of opinion that is intended to be current at the time of writing. It was our intention to minimize any potential bias by seeking the opinions of individuals who were recommended to us by industry, in addition to seeking the opinions of others who were also experts in their fields of research and northern Canada issues.

The diversity of the tasks and the enormity of the information base necessitated the focusing of the activities and inevitably not all information will have been assessed.

The co-operation of the diamond mining industry was required during the process of gathering and assessing information, and of seeking opinion regarding priority research activities. This co-operation was sought and obtained through meetings and correspondence, and the objectives of this study were also presented to the industry and to others at the Environment Canada sponsored workshop that was held in Ottawa in June 2003 (Intersol Consulting Associates Ltd. 2003). Co-operation was also received from representatives of the placer gold and oil sands mining industries.

Details regarding consultations and opinions that were obtained from a number of knowledgeable people have been placed in an appendix within a separate report (Samis et al. 2005). Reference to this body of information is also to be found in the main text of this document and in a summary report (Birtwell et al. 2005). The adopted approach provides an audit trail of activities, of opinion and of findings so that others may assess the value of the procedures and the information contained herein.

LITERATURE SEARCH AND FOCUS

The organization of the literature search was the initial step in gathering information. The search was focused upon topics that were considered to be relevant to the tasks, and these are indicated in the section headings below.

To simplify the tasks and place a priority on important factors, information on habitat that is considered to be critical to the survival of fish, and how they use that habitat was assessed. By adopting this approach we hoped to reveal the risks associated with the degradation or removal of this habitat due to mining. But, also of importance, was the identification of critical habitat for accommodation within compensatory and restorative measures as outlined in the Habitat Policy (Department of Fisheries and Oceans, DFO 1986). The initial search of the literature yielded approximately 3,000 documents. All records were managed using the Procite® software by ISI ResearchSoft (800 Jones Street, Berkeley, CA 94710, USA) which is compatible with Microsoft Windows.

Selected literature was assessed and conclusions drawn. Similar and parallel deductions about critical habitat were also made independently by those addressing "Species at Risk" (Randall et al. 2003). Thus a common need for the identification (and protection) of critical habitats has been clearly stated, notwithstanding the need for the protection of the function of them in watersheds and aquatic systems in general, and understanding the implications of reductions in habitat and the associated risks to aquatic communities.

Our intention was to reveal deficiencies in our knowledge at different levels of biological organization from the life stages of species to population levels, and habitat changes in watercourses and lakes, watersheds and larger drainage basins. This approach was guided by the premise that in order to understand more complex systems a suitable starting point is the simplest one that is available that contains all the parameters of interest (Platt 1964, cited in Johnson 2002). It is known that fish will use sub-optimal habitat (Birtwell and Korstrom 2002*a*; Birtwell et al. 1999, 2001*a*; Jones and Tonn 2004), but the challenge is to identify that which is optimal for ensuring the maintenance of fitness and performance of individuals. The reasoning behind such a position is the need to protect the health of organisms because of the competitive environment in which they exist. Fish that are stressed, debilitated, and conspicuous are rapidly consumed (Kruzynski et al. 1994; Mesa et al. 1994; Birtwell et al. 2001*b*). The typical absence of the corpses of the countless organisms that die naturally is testimony to the assimilative processes in aquatic systems wherein debilitated and dead organisms are consumed by predators and scavengers, and also through decay (Elliott 1997). While natural pristine habitat in northern locations may well be in "stable" equilibrium from an ecological perspective (Johnson 2002), human interventions (e.g. global warming) are changing this and the challenge will be to determine in what manner it is changing and to predict the consequences (Schindler 2001). By providing and protecting the function of optimal habitat one would expect the benefits to accrue, in turn, to maintaining the health of individuals, to stocks and to populations (Hayes et al. 1996).

REPORTING

The results of the literature search are contained within a separate report (Khan et al. 2005). Provision has been made to provide easy access to the information base through the use of data-sorting techniques, and the user-friendly CD version of the data fulfills that function.

By adopting the above-mentioned approach we hoped to reveal the risks associated with the degradation or removal of fish habitat due to mining. But also of importance was the identification of habitat structures and functions for accommodation in mitigation, compensatory and restorative measures. The consistent application of the *Fisheries Act* related to mining and habitat management was also examined.

Recommendations to address the deficiencies in knowledge and understanding, and approaches to decision making regarding the effects of oil, placer gold and diamond mining on fish and their habitat were deduced from the above-mentioned reviews and consultations. Figure 2 is a schematic diagram that reveals the linkages among components of this process.

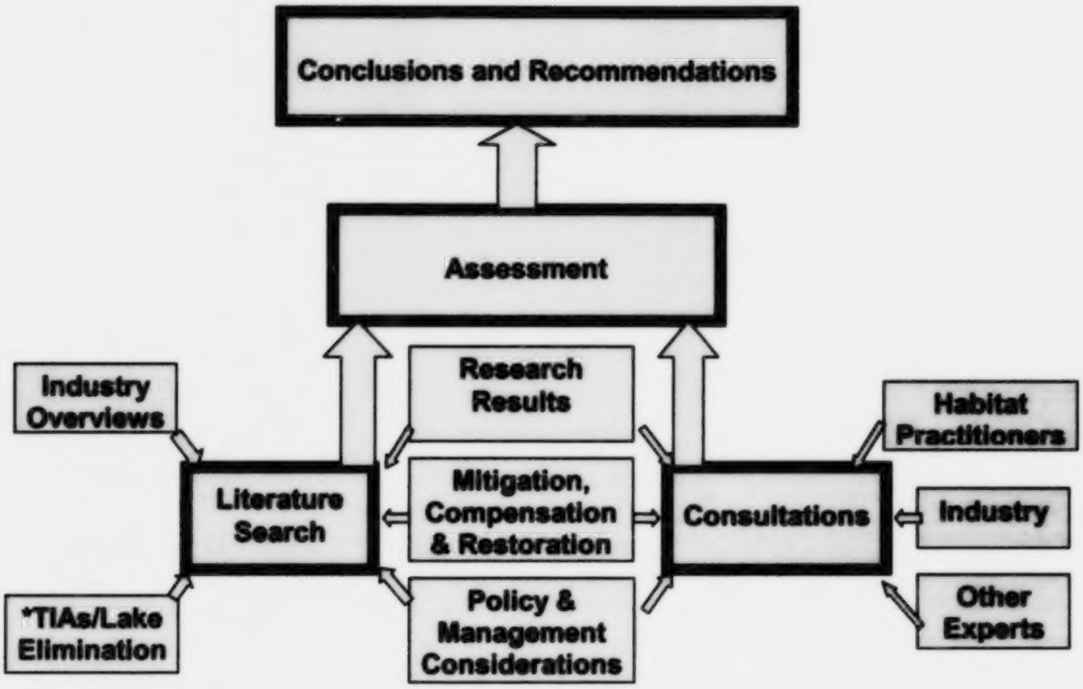

Conclusions and Recommendations
Assessment
Industry Overviews
Research Results
Habitat Practitioners
Literature Search
Mitigation, Compensation & Restoration
Consultations
Industry
*TIAs/Lake Elimination
Policy & Management Considerations
Other Experts

Figure 2. Main components of the tasks that were carried out in this review, their interrelationships, and the processes leading to the conclusions and recommendations. (*TIAs – tailings impoundment areas).

LITERATURE REVIEW

Seven species of fish were selected for this component of the literature review. They are commonly found in lakes and streams in the Lac de Gras area of the Canadian Barrenlands (Martin 2001), a region where diamond mining is currently focused. The seven species (lake trout, *Salvelinus namaycush*; round whitefish, *Prosopium cylindraceum;* lake whitefish, *Coregonus clupeaformis*; Arctic grayling, *Thymallus arcticus;* longnose sucker, *Catostomus catostomus;* burbot, *Lota lota;* slimy sculpin, *Cottus cognatus*) selected for this review are of both economic and ecological importance in the Canadian Arctic. They are generally sympatric and often occupy the same ecological niche.

The literature was searched for information on various aspects of the biology and functioning of these species, including distribution, habitat preferences and critical requirements, diet, seasonal changes, behavior, physiology, life cycle stages, residency and migratory behavior, species associations, limiting factors, tolerance, and resistance to stressors.

The information provided in this review is primarily based on peer-reviewed journal articles and also with reference to technical and manuscript reports. A number of reviews on the biology of freshwater fishes, in particular those by Richardson et al. (2001) and Evans et al. (2002) which specifically dealt with life history characteristics of freshwater fishes occurring in the Northwest Territories and Nunavut assisted in this process.

Similarly, reviews of freshwater fishes of Newfoundland and Labrador by Bradbury et al. (1999) and of the Peace, Liard and Columbia River drainages by Ford et al. (1995) also provided substantial information for this review. All reviews concluded that there is a significant lack of information on specific habitat requirements of Arctic fishes. In addition, a number of compilations of information on freshwater fishes of a more regional nature that have appeared since the publication of the comprehensive work on freshwater fishes of Canada by Scott and Crossman (1973) also provided useful information for the current review, for example, Lane et al. (1996*a*, 1996*b*, 1996*c*). These three documents are a compilation of habitat requirements for spawning, nursery, and adult life stages in lakes for fishes occurring in the Great Lakes; Portt et al. (1999) – a compilation of information on riverine habitat requirements for fish species of the Great Lakes; Portt et al. (1988) and Minns et al. (1993) – compilations of information relating to ecomorphometric and ecological characteristics of 75 species occurring in freshwaters of Ontario; Nelson and Franzin (2000) – descriptions of habitat availability and its utilization by 11 species of fish from the Assiniboine River, Manitoba; Roberge et al. (2001) – descriptions of life history characteristics of freshwater fishes of British Columbia with emphasis on lake habitat characteristics; and Coker et al. (2001) a compilation of information on morphological and ecological characteristics of Canadian freshwater fishes.

We have also relied on a number of reviews that have focused on the biology of individual species (such as Arctic grayling reviews by Armstrong (1982); Armstrong et al. (1986) and Northcote (1995), and a review of burbot by McPhail (1997)).

Very little information is available on the ecology, physiology and behavior of these species under Arctic conditions. Knowledge is scant or often lacking on the distribution, life history traits, and habitat requirements of various life stages. Accordingly, and where appropriate, reliance was placed on the literature on the selected species from more southerly locations in Canada and the United States, especially the Great Lakes Area. This review nonetheless focuses on the biology and ecology of the seven selected fish species under conditions of the Arctic environment, which pose different challenges to these species than the conditions in more southerly areas of their range. This is followed by a commentary on watershed ecology and cumulative effects of human interventions in the Canadian Arctic with particular reference to the Slave Geological Province and the Coppermine River watershed.

LAKE TROUT (*Salvelinus namaycush*)

The habitat and biological characteristics of lake trout are summarized in Table 1. Tables 2 and 3 provide a qualitative assessment of the state of knowledge about the biology of lake trout in the Arctic versus more southerly regions.

Distribution

Lake trout are distributed throughout North America including many Arctic islands (McPhail and Lindsey 1970; Khan and Qadri 1971; Scott and Crossman 1973; Gunn and Pitblado 2004). Although native to North America, this species has been successfully introduced in other parts of the world such as Sweden, South America and New Zealand. Lake trout have been successfully crossed with brook trout (*S. fontinalis*) to produce fast-growing and fertile hybrids called "splake" that are capable of maintaining self-sustaining populations in the Great Lakes (Berst et al. 1981). Both lake trout and splake are highly prized sport fish. In Canada's north, especially the NWT, there is a potential for expanding sport fishing for lake trout and prospects for commercial fisheries have also been considered (Yaremchuk 1986; Yaremchuk et al. 1989; Yaremchuk and Wong 1989). However, this latter opportunity is debatable for over-fished lake trout populations which have not shown a strong recovery from population declines in the past (exploitation in the early 20th century in Lesser Slave Lake; recovery has not occurred since the 1940s; Schindler 2001). In the southerly portion of their range, lake trout are commonly found in deep, clear, oligotrophic lakes. They are also common in relatively shallow isothermal lakes in the Canadian Arctic (Scott and Crossman 1973). While the lake trout is a typical lacustrine species, it is known to exhibit both lacustrine and adfluvial life history types and has been reported from large, clear rivers as well (McPhail and Lindsey 1970; Scott and Crossman 1973). However, it is to be noted that "early summaries of lake trout life history may not be correct because bull trout were considered a form of lake trout" (Ken Mills, Research Scientist, Environmental Science Division, DFO, Winnipeg, Manitoba; pers. comm.). Bull trout (*Salvelinus confluentus*) superficially look very similar to lake trout and are river dwelling and river spawning. It is, therefore, possible that adfluvial forms of lake trout reported by earlier authors were in fact bull trout.

Biology of life history stages

Adults: Adults generally inhabit deeper parts of lakes. Lakes containing lake trout are characterized by deeper, clearer, and colder waters. These lakes are also generally lower in total alkalinity, buffering capacity, total dissolved solids, and morphoedaphic index values than other lakes (Martin and Olver 1980). In a comparison of 2,500 Ontario lakes, Johnson et al. (1977) reported that lake trout were most frequently found in lakes characterized by a mean depth >6 m, total dissolved solids <50 mg/L, average hypolimnetic oxygen concentration >6 mg/L, and metric morphoedaphic values (mean depth/total dissolved solids) >6. In the north, however, where average summer water temperatures are lower, lake trout are known to inhabit shallow lakes (McPhail and Lindsey 1970; Ryan 1988). In sub-Arctic areas, lake trout concentrate on their spawning grounds along shorelines between mid August and October (Johnson 1972; Martin and Olver 1980). However, Johnson (1972) observed that preferences for shoreline or offshore areas of the lake were related to the size of lake trout for Arctic populations. He reported that large lake trout (>400 mm) were captured along shorelines and smaller lake trout (<400 mm) were captured in offshore areas. Adams (1997) reported similar findings in Walker Lake, Alaska (67°08'N, 154°20'W) where small lake trout constituted 92% of the fish captured at offshore sites and only 14% of captured fish at shoreline sites. Conversely, McDonald et al. (1996) stated that young lake trout occupy shallow habitats close to shore, and they considered that this behavior and habitat use was because of risk of predation. In such areas, however, they may encounter poorer feeding conditions than in offshore regions which ultimately may affect their survival. In oligotrophic Arctic lakes McDonald et al. (1996) considered that fish such as lake trout are at their physiological limits. Under these oligotrophic conditions food is likely to be limiting, so any perturbation that affects food (such as ice cover in shoreline areas) will have consequences to the consumers. In the same context, it has been postulated by Martin and Olver (1980) that large lake trout enter the shallow portions of a lake for feeding and their presence may cause small lake trout to move away from littoral areas and compete for food in the offshore areas.

Eggs: Eggs are laid on cobble and gravel substrates and they fall into the interstitial spaces in these substrates where they may be incubated for up to five months over the winter and early spring (Marsden et al. 1995*a*, 1995*b*). Substrate composition is an important factor for the survival and development of eggs because the size of cobble or gravel should be appropriate to secure eggs and at the same time allow water circulation around eggs to provide adequate dissolved oxygen (Marsden et al. 1995*b*). As such, the preferred spawning substrate is rounded or angular cobble exceeding 1-2 cm in diameter (Marcus et al. 1984; Thibodeau and Kelso 1990).

Incubation time varies considerably depending upon the latitudinal location and water temperature, which along with genetic origin of the stock, are factors affecting the development and survival of eggs (Martin and Olver 1980). The incubation period increases with decreasing temperatures (Garside 1959, quoted in Ford et al. 1995). According to Martin and Olver (1980), temperature tolerance of eggs is between 0.3 °C and 10.0 °C with an optimum incubation temperature of 5.0 °C. However, it has been noted that the 5.0 °C temperature does not typically occur in gravel used by wild spawning stocks (Ken Mills, Research Scientist, Environmental Science Division, DFO, Winnipeg, Manitoba; pers. comm.).

In shallow areas, ice contact with eggs can cause mortality of eggs (Sly and Schneider 1984). Lake trout eggs are also sensitive to sediment deposition, which often causes egg mortality (Sly and Schneider 1984).

Larvae and juveniles: Eggs usually hatch in March or April in the Great Lakes area, but in northern areas, such as Great Bear Lake, hatching may occur as late as June (Scott and Crossman 1973; Marsden et al. 1995*a*, 1995*b*).

Information on post-hatching movement and distribution of lake trout fry is limited, probably as a result of the difficulty in capturing these very small fish (Ford et al. 1995). Earlier authors (Royce 1951; Martin 1957; DeRoche 1969, cited in Ford et al. 1995) have reported that fry move out of the spawning area into deeper waters soon after yolk sac absorption. However, some authors (Scott and Crossman 1973; Peck 1982) have observed that after emergence, fry remain in spawning areas for several weeks to several months, feeding on insects and crustaceans. The diet changes as the juveniles grow and may include small fishes (Scott and Crossman 1973).

Newly-emerged fry prefer a water temperature of approximately 12 °C (McCauley and Tait 1970; Peterson et al. 1979), but juveniles move to deeper areas as water temperature exceeds 15 °C (Peck 1982).

Juveniles (young of the year (YOY) to 3-4 years of age) inhabit cool, deep waters of oligotrophic lakes during periods of summer stratification in the southern part of their range (Martin and Olver 1980). Direct observations by using a remotely-operated vehicle in Lake Opeongo and Source Lake in Algonquin Park, Ontario suggested that juvenile lake trout moved to shallower habitat at night (Davis et al. 1997). These authors observed the highest densities of juveniles during the day at 15-20 m depth, and at night the highest densities were at depths between 5 and 10 m.

Young lake trout are solitary rather than schooling and tend to stay very close to the lake bottom, usually <0.3 m from lake sediments (Davis et al. 1997).

Low levels of dissolved oxygen (2.6 to 4.5 mg/L) have adverse effects on larval development such as retardation in development rates (Garside 1959, quoted in Ford et al. 1995).

Ecology of life history stages

Habitat use: Habitat needs of lake trout, like other salmonids, vary with season and stage of life cycle (Bjorn and Reiser 1991). Being a cold water fish, habitat selection and use by lake trout is highly dependent on water temperature and oxygen. Both these factors could be potentially lethal at their extremes (Ryder and Edwards 1985). Thus, some authors (such as Ryder and Kerr 1989; Ryan and Marshall 1994) have provided evidence to support the notion that "if a lake is to support a self-sustaining population of lake trout, it must be suitable with respect to the primary environmental determinants (dissolved oxygen, water temperature, subsurface light, and dissolved nutrients) that control or limit the species" (Marshall 1996). In the southern range of its distribution, lake trout prefer cool (about 10 °C) deep lakes, but in the north (such as tundra regions) where ambient temperatures are lower, they also inhabit shallow lakes (McPhail and Lindsey 1970; Ryan 1988).

According to Carl et al. (1990), regardless of season, juvenile lake trout have generally been reported to occur in deeper waters than adults in Ontario lakes and yet prefer temperatures between 10 and 16 °C. However, it has been shown by McDonald et al. (1992) that YOY lake trout may use inshore habitat with suboptimal conditions for growth to avoid predation by larger fish. There are few reports of habitat use and preferences of very young lake trout and this is an obvious area requiring future research.

Spawning: In most areas of Canada, spawning takes place in late summer to early fall (McPhail and Lindsey 1970; Scott and Crossman 1973). In northern locations such as the Northwest Territories, however, lake trout spawn mainly in the fall (Scott and Crossman 1973). It is generally held that spawning is triggered by declining water temperatures and photoperiod (Martin and Olver 1980).

Although lake trout are essentially a lake-spawning species, several authors have reported evidence of river-spawning populations of lake trout. River spawning of lake trout has been reported from eastern Lake Superior (Loftus 1958), Lake Nipigon (Dymond 1926, cited in Evans et al. 2002), and Cedres Brook, Quebec (Martin and Olver 1980). There are unconfirmed reports of river spawning of lake trout in certain rivers of Alaska (Morrow 1980). There are also anecdotal reports of river spawning in the wide and slow sections of the Back River in Nunavut. Although, lake trout are reported to occur throughout the northern river systems in Canada, there are no reports of distinct river-dwelling populations in Canada (Scott and Crossman 1973). However, information is lacking with respect to the extent and nature of use of the riverine habitat, and it is possible that some of the reports of riverine lake trout may have been confused with bull trout (Ken Mills, Research Scientist, Environmental Science Division, DFO, Winnipeg, Manitoba; pers. comm.).

Lake trout are iteroparous (spawn once annually or produce eggs in successive years) but they may not spawn each year. In Great Slave and Great Bear Lakes, for example, spawning appears to take place every second or third year (McPhail and Lindsey 1970). Spawning takes place at night (Gunn 1995) in late summer or fall usually in shallow areas of lakes and primarily over cobble, rubble, and large gravel substrates, interspersed with boulders (McPhail and Lindsey 1970; Scott and Crossman 1973). These areas are generally free of sand, silt, clay and mud (Marsden and Kruger 1991; Kelso et al. 1995) and are often characterized by wave action and water currents which minimize deposition of sand and organic material in interstitial spaces (Sly and Evans 1996). However, lake trout have been reported to have spawned over sand, silt and mud substrates (Goodyear et al. 1982; Beauchamp et al. 1992, cited in Northcote 1995).

Spawning of lake trout is reported to generally occur at depths from 0.12 m to 55 m (Scott and Crossman 1973), but spawning at depths in excess of 100 m has also been reported (Thibodeau and Kelso 1990). In smaller northern lakes, lake trout spawning has occurred in waters <5 m deep (Marsden and Krueger 1991).

In their review of the literature on lake trout spawning habitat in the Great Lakes, Marsden et al. (1995*a*) concluded that "the selection of spawning sites by adult lake trout may be influenced by a variety of factors, including olfactory cues (homing), visual cues (recognition of substrate type), reef location with respect to shoreline, water depth, proximity to deeper water, reef size and contour, substrate size and shape, depth of interstitial spaces, water temperature at spawning time, water quality in interstitial spaces, and the presence or absence of other species." These factors are briefly discussed below.

Homing: Although the role of homing in lake trout spawning is unclear (Marsden et al. 1995*b*), there is evidence that native lake trout stocks in the Great Lakes return to their natal spawning areas (Schram et al. 1995). There are also laboratory data suggesting that spawning lake trout are attracted by odors associated with early life stages (Foster 1985). However, Marsden et al. (1995*a*) have demonstrated in field experiments that lake trout in the Great Lakes preferred artificial substrates composed of large cobble or rubble devoid of organic debris over natural substrates that are filled with organic debris. They, therefore, suggested that while homing may be important in the selection of spawning habitat by adult lake trout, it may not be essential for finding suitable spawning habitat. These authors have argued that habitat quality as "perceived" by adult lake trout may be a more important factor in spawning habitat selection than familiarity with the site utilized for spawning in previous years (Marsden et al. 1995*a*). This argument by Marsden et al. (1995*a*) therefore, suggests that substrate size and shape, as well as interstitial space are critical for spawning site selection by adults and protection and survival of eggs and fry.

Spawning site and substrate: In the Great Lakes, lake trout are reported to spawn in shallow as well as deep waters, at both nearshore and offshore sites (Brown et al. 1981; Goodier 1981). The relative success of spawning at these sites in terms of contribution to year class recruitment, however, is unknown (Eshenroder et al. 1995; Marsden et al. 1995*b*).

Lake trout are reported to deposit eggs on sites varying in area from a few hundred m^2 to several km^2 (Fitzsimons 1995; Schreiner et al. 1995). In extreme cases rock piles less than 5 m in diameter were utilized for egg deposition (Marsden et al. 1995*b*). Some authors have provided evidence that the contour of the spawning site is an important factor in site selection by lake trout. Steep slopes at spawning sites of lake trout are common in Great Lakes (Marsden and Krueger 1991; Casselman 1995; Fitzsimons 1995; Kelso et al. 1995). It has been hypothesized that the prevailing stronger currents at these steeply sloping lake beds help maintain good water quality (e.g. dissolved oxygen) for egg incubation in interstitial spaces (Marsden et al. 1995*a*).

Optimum water quality at spawning sites: The optimum water quality requirements for lake trout egg development are not well known (Sly 1988). It has been proposed that long-term monitoring over large areas is needed to adequately characterize water quality because of the dynamic nature of biological and meteorological processes that affect it (Sly 1988; Manny et al. 1989; Perkins and Krueger 1995; Marsden et al. 1995*a*). Several authors have provided evidence to show that timing of spawning in the Great Lakes is partly dictated by the seasonal decrease in epilimnetic water temperatures to 12 °C (Casselman 1995; Marsden et al. 1995*a*, 1995*b*; Fitzsimons 1995).

McAughey and Gunn (1995) experimentally tested the behavioral response of lake trout to the loss of spawning habitat in a 67-ha lake in Ontario. When 15% of traditional spawning grounds were covered by plastic tarpaulins and mesh fencing to prevent spawning lake trout from utilizing the site, the fish responded by using new sites around the lake. Lake trout responded in a similar manner after the additional removal of spawning sites the following year. The intentional removal of spawning habitats in this study produced no detectable change in timing of spawning and fish abandoned the traditional sites. Whether the results of these experiments are applicable in other lakes and areas has yet to be determined.

Sly and Evans (1996) have provided a review of literature relating to the reproductive ecology and suitability of habitat for spawning lake trout with reference to Ontario lakes Simcoe and Manitou, and Seneca Lake (New York). These authors provided a dichotomous key for determining the optimum spawning habitats necessary in restorative or habitat compensatory activities. An important deduction of this review was that spawning lake trout commonly used stable "lag" deposits derived from glacial sediments or relict features such as fans, bars, or submerged talus slopes. Artificial breakwaters of broken material may also provide suitable substrates. While the optimal particle size of substrate ranges from 4-10 cm in diameter in these lakes, larger materials up to 30 cm have also been utilized for spawning by lake trout in some areas. Because the transport of finer sedimentary materials can limit the suitability of some substrates, spawning sites were usually away from depositional areas. Water quality for successful embryo development was characterized by dissolved oxygen concentrations above 7 mg/L and low un-ionized ammonia (<12.5 µg/L) in the interstitial waters of spawning habitats.

Rearing: Information on the movement of newly-hatched lake trout fry to appropriate rearing and nursery areas is limited, presumably owing to a lack of suitable sampling techniques for the elusive early stages in the first year of a lake trout's life

(Marsden et al. 1995*a*, 1995*b*). There is a need to develop improved methods to sample early stages of lake trout (and other species).

Studies in Lake Superior suggested that newly-hatched fry move with downward currents from the spawning shoal (Bronte et al. 1995). However, it is not clear if this represents a directed movement or passive drift. In one stream-spawning population, it was observed that the hatched fry moved directly towards their nursery lake (Patterson 1968, cited in Ford et al. 1995). In a lake-spawning population in Lake Superior, however, Peck (1982) reported that after hatching, lake trout fry in the vicinity of a shallow water spawning reef stayed for up to three weeks before moving into deeper waters. It has been suggested that this period is probably important in olfactory imprinting before dispersal into the hypolimnion (Horrall 1981, cited in Ford et al. 1995). In Lake Superior, after the initial inshore residence period, lake trout fry have been found in depths of less than 37 m (Eschmeyer 1956, cited in Ford et al. 1995). In other lakes such as Lake Tahoe in California, they have been observed in depths greater than 100 m (Martin and Olver 1980).

The preferred temperature for the newly-hatched fry has been reported to be about 12 °C (McCauley and Tait 1970; Peterson et al. 1979, cited in Ford et al. 1995). Casselman (1995), in both laboratory and *in situ* experiments, reported that in eastern Lake Ontario spawning has been observed at a mean water temperature of 11.5 °C (range 12.7 °C in mid October and 8.8 °C in mid November). Casselman's experiments suggested that spawning at lower temperatures would increase fry production and that temperature inversely affects fry survival, especially if the spawning substrate is degraded by organic sedimentation which causes increased biochemical oxygen demand (BOD) and reduced dissolved oxygen levels. Typically, YOY lake trout leave inshore areas when water temperature exceeds 15 °C (Peck 1982), however, predation pressure may result in occupancy of these less-productive regions of Arctic lakes (see below).

Adults leave the spawning grounds shortly after spawning (Scott and Crossman 1973; Morrow 1980).

Food supply: Lake trout are omnivorous and display marked seasonality in their diet (Martin 1952). Several authors have provided evidence of considerable foraging flexibility which may involve piscivory, benthivory and planktivory (Martin 1952, 1966, 1970; Konkle and Sprules 1986; Trippel and Beamish 1989, 1993; France and Steedman 1996). Seasonal changes in diet are likely caused by the availability of different food items through the year as observed by Martin and Olver (1980). These authors observed that with the increase in surface-water temperatures, juvenile lake trout move into deeper waters and, as such, shallow-water invertebrate species disappear from diet and are replaced by deeper-water benthic organisms and fishes. In the fall, the diet again reverts to the shallow-water forage organisms as the temperature drops and the juveniles move to shallower areas for feeding, and predator avoidance (Martin and Olver 1980).

A 1978-81 survey (Elrod 1983) of the food of juvenile lake trout in the US waters of Lake Ontario showed that invertebrates were an important food for lake trout <200 mm in length, but were only occasionally eaten by larger lake trout. For all seasons and size groups of juvenile lake trout combined, slimy sculpin was the principal (42% by weight) forage fish species consumed. The percentage composition of forage fish in lake trout diet varied from season to season. When comparing the results obtained from Lake Ontario with information from other lakes, Elrod (1983) found that the diet of lake trout varied significantly between lakes and between seasons within a lake. In a 1973-1982 survey of diet and selection of major prey species by Lake Michigan salmonines, Jude et al. (1987) concluded that there was no significant difference in proportion of feeding fish among seasons. The total weight of prey was almost entirely of fish, but some invertebrates were eaten in spring and summer.

Planktivorous foraging in lake trout is largely caused by an absence of forage fish as a primary source of food (Konkle and Sprules 1986). Foraging on plankton causes stunting in lake trout, and the population becomes dominated by slow growing or stunted individuals (Martin 1966; Konkle and Sprules 1986). However, when stunted, planktivorous lake trout are transferred to lakes where forage fish are available, the lake trout change their diet to fishes, grow faster, and live longer (Martin 1966). Konkle and Sprules (1986) demonstrated that stunted lake trout in Algonquin Park lakes in Ontario were facultative planktivores in summer. These fish mainly fed on zooplankton although other prey such as mayfly larvae and cyprinids were present in those lakes. Metabolic demands of foraging on small and dispersed zooplankton appear to offset the potential advantage of exploiting the relatively large energy source provided in the zooplankton (Kerr and Martin 1970).

The omnivorous nature of lake trout and their foraging flexibility, as shown by stable isotopes, was demonstrated by France and Steedman (1996). Their study suggested that juvenile lake trout in northwestern Ontario lakes were relatively omnivorous, "exhibiting individual $\delta\ ^{15}N$ values (deviations of $^{15}N/^{14}N$ ratio from the isotopic standard) consistent with predation on opossum shrimp *Mysis relicta* (41% of samples), zooplankton (35% of samples), and littoral organisms (25% of samples)." The carbon isotope ratios indicated that juvenile lake trout in northwestern Ontario lakes obtained about half of their carbon from the littoral (i.e. outside the deepwater pelagic) zone. This ability of lake trout to consume both littoral and pelagic organisms suggests certain degree of resilience to possible food web alterations by human interventions. This resilience was demonstrated when extirpation of *M. relicta* (an important food of lake trout) in experimentally-acidified lakes (Nero and Schindler 1983) did not cause a collapse of a resident lake trout population, and only resulted in lower condition factors as lake trout switched to a diet of littoral cyprinids (Mills et al. 1987). In these experiments, the condition of lake trout did not collapse until all prey species disappeared or had severely-reduced abundances. However, the condition of lake trout rebounded during pH recovery immediately after two of the original 5 forage species resumed reproduction and produced small YOY fish (Mills et al. 2000).

Migration: Lake trout have been determined to make extensive migrations. While the home range of lake trout is relatively small, they have been reported to move distances of 300 km in a period of less than one month in Lake Superior (Eschmeyer et al. 1952). In Great Slave Lake, on the other hand, Keleher (1963) reported stocks that did not migrate and remained in their natal habitat or a limited part of the lake throughout their lives. At the Gull Island Shoal (Lake Superior), the age 0 lake trout have been reported to move 7-11 km from hatching to nursery grounds over a period of 3 months (Bronte et al. 1995).

Stream and river spawning in lake trout is rare but it is reported to occur in some Arctic rivers (MacDonald and Stewart 1980, cited in Evans et al. 2002) and the Laurentian Great Lakes (McPhail and Lindsey 1970). In eastern Lake Superior, lake trout spawn in six different tributaries and have been reported to migrate distances from 180 m to 3.2 km before reaching the riverine spawning grounds (Loftus 1958, cited in Ford et al. 1995).

Refugia: The movement of lake trout is related to variations in surface water temperatures, and as such reflects adaptive thermoregulatory behaviors. In summer, after thermal stratification, lake trout are reported to move below the thermocline (McPhail and Lindsey 1970). In shallow Ontario lakes which become isothermal in summer (15 ^{o}C to 20 ^{o}C), lake trout have been observed to congregate in cool groundwater discharges when ambient temperature exceeds their tolerance range (6 ^{o}C to 13 ^{o}C) (Snucins and Gunn 1995).

These groundwater refugia in relatively shallow lakes need to be protected as critical habitats for lake trout.

Behavior

Feeding and predator avoidance: The eggs and fry of lake trout are vulnerable to predation by other fishes. In the Great Lakes, lake trout predators include both indigenous species and recent invaders such as alewife (*Alosa pseudoharengus*). In the Arctic, however, the number of predators is lower because of low biodiversity and a lack of exotic invaders. The potential predators of lake trout eggs and fry in Arctic lakes and rivers would be slimy sculpin (*Cottus cognatus*), burbot (*Lota lota*), round whitefish (*Prosopium cylindraceum*), lake whitefish (*Coregonus clupeaformis*), lake chub (*Couesius plumbeus*), longnose sucker (*Catostomous catostomus*), seven-spine stickleback, and other lake trout. In the Great Lakes, all these species or close relatives have been reported to prey on lake trout eggs and fry (Jones et al. 1995).

Since little is known about predator-prey relationships of these species in the Arctic, research in this direction would generate useful information for fisheries and habitat management purposes. In particular, the predator-prey relationship between slimy sculpin and age 0 lake trout needs attention because their habitats overlap spatially on reefs and on lake trout spawning areas.

Lake trout are particularly vulnerable to predation by benthic predators (such as sculpins and burbot) as sac fry move out of the spawning substrate to feed or to fill their swimbladders with air (Baird and Krueger 2000).

The interaction of lake trout with slimy sculpin (*Cottus cognatus*) on reefs and adjacent nursery areas of lake trout can play an important role in the recruitment and ultimately the population dynamics of lake trout (Hudson et al. 1995). The interactions resulting from such overlap include predation on each other's eggs. In these areas slimy sculpins overlap with age 0 lake trout. Thus, in lakes with higher sculpin to lake trout ratios, sculpin predation on age 0 lake trout could be a "bottleneck" for lake trout recruitment (Hudson et al. 1995).

Owens and Bergstedt (1994) in studying the 1986-1987 abundances of slimy sculpin and juvenile lake trout in Lake Ontario found that abundance and biomass of slimy sculpin declined and those of juvenile lake trout increased at depths less than 70 m, a depth where the populations of the two species overlap. The authors did not find any significant negative correlations between abundance of slimy sculpin and those of the other two most abundant species (alewife and rainbow smelt). Because slimy sculpins are the principal fish eaten by juvenile lake trout at depths less than 70 m, the authors concluded that lake trout altered the slimy sculpin population. The authors also suggested that areas deeper than 70 m also serve as refuges for slimy sculpin.

Biga (1995) and Biga et al. (1998) studied the relationship of rock size and the size of mottled sculpin (*Cottus bairdii*) to assess the role of these two variables in the predation of this species on lake trout and rainbow trout eggs. They found that sculpins collected from rubble (10 to 20 cm glacial till) substrate were larger and had more lake trout eggs in their stomachs than the sculpins collected from gravel substrate (5 to 10 cm). They also found that substrate size can minimize predation pressure on lake trout eggs by excluding larger sculpins from interstitial spaces where the eggs tend to be found. A modeling study (Savino et al. 1999), designed to examine the effects of multiple predators on survival of eggs and fry of lake trout, predicted that interstitial predation on eggs accounted for most (76 to 81%) of the predation on early stages of lake trout, and epibenthic egg predation (12 to 19%) and fry predation (0 to 12%) had less effect on lake trout survival. This study also estimated that with a starting egg density of 100 egg/m^2 and with initial predation conditions (2 eggs/m^2) no lake trout would survive to swim-up stage, but at egg densities of 250 eggs/m^2 and 500 to 1,000 eggs/m^2 the survival would be 36% and 70 to 80% respectively.

The YOY lake trout are known to be almost exclusive feeders on zooplankton and aquatic insects (Martin and Olver 1980). However, as they grow older, they undergo ontogenetic diet changes to larger aquatic invertebrates and fishes. But competition with larger fish may jeopardize this dietary shift for optimal growth (Martin and Olver 1980). YOY lake trout are subjected to predation by larger fishes including other lake trout. Johnson (1972, 1976) reported a bimodal size distribution of lake trout populations in the Canadian Arctic and suggested that this pattern of distribution was the result of larger lake trout forcing smaller lake trout into suboptimal habitat to avoid predation. It has been hypothesized by McDonald et al. (1992) that YOY lake trout may use inshore habitat with suboptimal conditions for growth to avoid predation by larger fish. To test this, these authors penned YOY lake trout in nearshore and offshore pelagic areas of two lakes: one with a lake trout population (Toolik Lake) and the other without lake trout. The results showed that YOY in the lake with trout lost weight but those offshore lost less weight. On the other hand, the YOY lake trout in the lake without lake trout gained weight and those in offshore areas gained more weight. Bioenergetics modeling of YOY lake trout suggested that the limited availability of zooplankton in the nearshore and offshore pelagic zone of Toolik Lake (lake with lake trout) would likely be insufficient to allow successful recruitment.

There is some evidence that lake trout predation on certain benthic invertebrates controls their population and community dynamics. For example, in Arctic Alaskan lakes the density and size distribution of the snail (*Lymnea elodes*) appears to be greatly affected by the presence or absence of lake trout (Hershey 1990; Merrick et al. 1991, 1992). A survey of 11 lakes in Alaska revealed that adult *Lymnea elodes* grew to a smaller mean size in lakes with trout than in lakes without trout (Hershey 1990). This study also suggested that the population dynamics of the snail *Valvata lewisi* which coexists with *L. elodes* in the lakes surveyed is controlled by lake trout predation rather than interspecific competition. Similarly, Goyke and Hershey (1992) in a study on the effects of fish predation on larval chironomid communities found that presence or absence of fish predators in tundra lakes in Alaska had a profound effect on chironomid community structure. They analyzed chironomid communities in systems with three types of predator regimes to determine the hierarchical effects of predation: ponds without fish present, lakes with slimy sculpin alone, and lakes with slimy sculpin along with lake trout and burbot. The results showed that lakes with burbot and trout along with sculpin had significantly higher chironomid biomass, density, richness and diversity than ponds without fishes present or lakes with sculpin as the only fish species. The authors attributed this effect to increased complexity of the predator regime. Ponds without fish had a greater percentage of predaceous chironomids most likely as the result of a lack of fish predation on these relatively large predaceous chironomids.

Migration: Bronte et al. (1995) studied hatching, dispersal and bathymetric distribution of age 0 lake trout at the Gull Island Shoal Complex in western Lake Superior. Catch data in this study suggested that age 0 lake trout moved 7-11 km from hatching to nursery grounds over a period of 3 months. In the nursery area, the bathymetric distribution of age 0 lake trout in July was bimodal with higher densities at the 15-19 m and the 35-39 m depth intervals than at other depths. Densities of fish gradually increased, and in August densities had increased two-fold but most fish were caught at 15-19 m. Towards the end of September, the densities were 10 times greater than in August, but most fish were captured at 30-40 m depth. Temperature profile data showed that age 0 lake trout were exposed to a range of temperatures, from 14 ^{o}C in late July to around 5 ^{o}C in September and early October.

The study by Bronte et al. (1995) also suggested that sand is a preferred substrate of lake trout in the nursery area, as their study area (Michigan Island, Lake Superior) is composed mostly of sand. This observation appears to be consistent with those of other authors. For example, Peck (1982) observed that age 0 lake trout moved from an artificial spawning area to sand substrate between May and August. Similarly, Eschmeyer (1956) had reported capture of lake trout fry at a sandy area adjacent to spawning grounds in central Lake Superior. Bronte et al. (1995) argued that the preference for sandy substrate by young lake trout is probably associated with the presence of *Mysis* sp. (an important food for young lake trout) on sandy substrates.

In a laboratory study, Ferguson et al. (1983) and Ferguson and Noakes (1983) observed that juvenile lake trout consistently displayed low agonistic behavior in response to varying water velocity, fish density and food ration, and spent more time close to the substrate with increasing water velocity. This restricted behavioral plasticity in lake trout suggests that its survival would, according to the researchers, be restricted to a limited range of habitats (Davis 1975; Ferguson et al. 1983; Ferguson and Noakes 1983).

An introduced population of lake trout in Lake Tahoe in the Sierra Nevadas, California has demonstrated an interesting behavior. It has been reported to spawn on deepwater mounds (40-60 m deep) over the macrophyte (*Chara delicatula*) (Beauchamp et al.

1992). The temperature (5-9 °C) and dissolved oxygen (7.2–8.8 mg/L) in this location at the time of spawning were considered to be favorable for egg incubation. Lake Tahoe is ultra-oligotrophic with Secchi depths ranging from 16 to 30 m (Goldman 1988; Loeb and Hackley 1988, cited in Beauchamp et al. 1992). This is an unusual spawning substrate for lake trout, but these authors observed that egg predators were very rare in the vicinity of the macrophyte beds and suggested that the mounds were refuges from egg predators.

Physiology and biochemistry

Growth, fitness, survival: It is well established that fish that forage on larger prey have lower energy expenditures associated with foraging than fish feeding on smaller prey (Martin and Olver 1980; Sherwood et al. 2002*a*, 2002*b*). When an adequate supply of forage fish is available throughout the year, adult lake trout are essentially piscivorous and, as such, demonstrate relatively rapid growth (Martin 1966, cited in Pazzia et al. 2002). However, in lakes that lack pelagic fish prey, lake trout are known to prey on littoral minnows after the fall overturn and during winter (Pazzia et al. 2002).

The reduced prey density in the littoral zone can prompt lake trout to switch to zooplankton and benthic invertebrates (Casselman and Grant 1998; Pazzia et al. 2002). Altered densities of preferred forage organisms are known to cause dietary shifts in lake trout and other fish and can have serious implications for energy requirements of lake trout populations (Sherwood et al. 2002*a*, 2002*b*).

Growth rates of lake trout in Lake Superior (with the lowest prey density) were lower than in Lakes Michigan and Ontario where there was high prey density (Eby et al. 1995). However, consumption rates (based on bioenergetics models) of lake trout were similar in these three lakes because the primary prey in Lake Superior (rainbow smelt, *Osmerus mordax*) have lower energy content than the primary prey (alewife, *Alosa pseudoharengus*) in Lakes Ontario and Michigan.

Stunting of lake trout populations in North American lakes has been reported and has been attributed to the absence of preferred food items (Martin 1966; Konkle and Sprules 1986; Donald and Alger 1986). Donald and Alger (1986) reported a stunted and slow-growing lake trout population from a subalpine lake in Jasper National Park, Alberta and attributed it to the absence of preferred lake trout food such as amphipods and other fish species in that lake. Similarly, Konkle and Sprules (1986) reported a stunted lake trout population in Lake Louisa, Algonquin Park, Ontario and suggested that the population was facultative planktivorous in summer, although other prey such as cyprinids and mayfly larvae were eaten, lake trout in this lake fed mainly on zooplankton, in particular large cladocerans (*Daphnia* sp. >0.9 mm) which formed about 99% of their diet. Martin (1966) provided evidence that planktivorous lake trout grow more slowly, reach smaller adult sizes, and mature at smaller sizes and earlier ages. Clearly, stunting in planktivorous lake trout represents a classic case where metabolic demands of foraging for small and disperse diet items (zooplankton) offset the potential advantage of exploiting the relatively-large energy reservoir of primary consumers (Konkle and Sprules 1986; Donald and Alger 1986). For example, when stunted lake trout were transferred from Lake Louisa to Lake Opeongo (where fish prey were abundant), they changed their diet to fish and grew faster and lived longer (Martin 1966).

Lake trout, like many aquatic consumers, have flexible feeding habits and, as such, diet and trophic position (i.e. position in the food chain) can vary both within and among populations (Martin 1966, 1970; Vander Zanden et al. 2000). Analyses of both stable isotopes and published dietary data by Vander Zanden et al. (2000) showed that lake-to-lake differences explained 78% of the total variation in lake trout trophic position. These results were consistent with earlier studies (Martin 1970) which concluded that among-population trophic differences largely determine patterns in the growth and life history of lake trout.

Burr (1997) in his study of 11 Alaskan lakes (9 sub-Arctic and 2 Arctic) demonstrated that the complexity of the fish community, was an important factor influencing lake trout growth parameters such as maximum size, age at maturity, and population density. In these lakes, asymptotic length varied from 517 to 984 mm and was positively correlated with length at 50% maturity, lake surface area, and fish species richness. Adult density was inversely related to lake surface area and to fish species richness.

In a comprehensive review of lake trout catch data from Lake Ontario, Carl et al. (1990) concluded that higher standing stocks of lake trout tend to occur in small lakes with few or no other fish species present due to lower trophic level transfer costs. On the other hand, it was shown that lakes with coregonines as prey produced larger lake trout due to the bioenergetics advantage provided by large prey size, but provided lower standing crops because of increased trophic level transfer costs, compared to lakes with only invertebrates for foraging by lake trout. The availability of prey fish species affected lake trout growth and the presence of other predators negatively affected the relative abundance of lake trout. Other measures of the aquatic community such as presence of *Mysis relicta* (major component of the diet of benthic-feeding lake trout), zooplankton size structure (major food of YOY lake trout), and fish size frequency distribution need to be closely examined for relevance in explaining variations in lake trout population parameters.

Growth rate in lake trout has also been reported to vary among age classes and stocks (Eschmeyer 1956; Healey 1978). Eschmeyer (1956) reported that while in some cases the growth rate is faster in smaller fish than in larger fish, the reverse is true in other cases. Healey (1978) showed that growth rate of lake trout varies between age classes and among stocks by up to a factor of two, but pointed out that while the sources of these variations may be genetic and environmental, other factors such as sampling and ageing techniques may also add to these variations. Martin and Olver (1980) have reported that growth rate may also be affected by the onset of sexual maturity.

An important determinant of growth rate is temperature, as it is well known that lake trout in relatively northern latitudes grow slower than in southern areas (Rawson 1961; Kennedy 1954; Eschmeyer 1956).

In Arctic lakes, large lake trout often greatly outnumber smaller lake trout and it has been hypothesized that smaller lake trout are excluded from open water habitat due to the presence of large piscivorous lake trout (Lienesch et al. 2001). Others, however, argue that "this is due to low annual recruitment, high annual survival, and growth slowing as fish become older" (Ken Mills, Research Scientist, Environmental Science Division, DFO, Winnipeg, Manitoba; pers. comm.).

Age at maturity: Age at maturity varies considerably, and in the Great Lakes lake trout have been reported to be mature at about 13 years (Scott and Crossman 1973). However, lake trout in northerly locations may mature at an earlier age. For example, in Great Slave Lake some individuals have been reported to mature as early as age 5 but most mature by age 11 (McPhail and Lindsey 1970). However, Bond (1975*a*, 1975*b*) reported that in some areas of the Northwest Territories, maturity may not occur before reaching age 19. Generally, males tend to mature at an earlier age than females. The onset of sexual maturity appears to be related to size because larger fish are reported to mature earlier than the smaller members of a population (Kennedy 1954; Rawson 1961, cited in Ford et al. 1995). It has been speculated that environmental factors such as light regimes may also influence age at maturity (Royce 1951, cited in Ford et al. 1995; Martin and Olver 1980).

It should be noted that difficulties with ageing lake trout have been reported by many authors (Scott and Crossman 1973; Healey 1978; Martin and Olver 1980). In older fish, there is little correlation between age estimated using otoliths and fish scale circuli. Also, it appears that the number of discernible circuli on lake trout scales is a function of fish length. Historically, scales were the first structures used to age lake trout and more recently, otoliths and other bony structures have been used. Therefore, there is great danger in comparing age at maturity and other parameters based on ages determined using different techniques.

Tolerance and resistance: Habitat modifications and destruction can stress lake trout. These modifications include the destruction of spawning and nursery grounds, blockage of migratory corridors, deterioration of water quality, reduction of oxygen in the hypolimnion and interstitial spaces of spawning substrates, and toxic effects of sediment-associated metals (Machniak 1975*a*; Evans et al. 1991; Fitzsimons 1995; Zint et al. 1995; Miller and Schram 2000). In a review of mitigation and compensatory strategies relating to lake trout habitats, Ford et al. (1995) pointed out that "impacts to spawning and incubation habitat have the most significant effects on lake trout populations." Evans et al. (1991) reviewed the effects of anthropogenic stressors on lake trout populations in Ontario lakes. These authors emphasized that the most direct effect of shoreline manipulation was on spawning substrates whereby fine inorganic material can clog interstitial spaces and smother eggs. Transport of sediment with organic material to lake trout spawning areas can also deplete oxygen in interstitial spaces to a level which jeopardizes survival of developing embryos (Sly 1988).

In western Lake Superior, although the red clay turbidity did not directly influence survival of sensitive stages of fish, it caused significant changes in the quality and intensity of light, even at low turbidity levels (Swenson 1978). These changes evoked behavioral responses in the fish community and, as such, influenced predator-prey relationships.

Increased levels of acidity can be detected by early life stages of lake trout, but they do not tend to avoid acidic waters (Gunn et al. 1987*a*, 1987*b*). Over-winter drawdown (via direct exposure of eggs) as well as recharge (via drowning of spawning habitat) can harm lake trout. If a drawdown reduces the availability of cold-water habitat (critical for lake trout survival) at depths that are subject to dissolved oxygen depletion the survival of lake trout may not be possible (Ryder and Edwards 1985).

Temperature tolerance of lake trout appears to depend on geographical location. For example, the temperature range for juvenile lake trout is between 6 °C and 13 °C (Martin and Olver 1980, but they have been found in temperatures from -0.8 °C in Arctic Canada (Boulva and Simard 1968) to 18 °C in Ontario (Martin 1952; Ford et al. 1995). A dissolved oxygen level of 4 mg/L is generally considered to be the minimum acceptable level for lake trout (Martin and Olver 1976; Davis 1975; Ford et al. 1995) and concentrations below 3 mg/L have been reported to be lethal (Patterson 1968). Serious developmental abnormalities in lake trout embryos were observed when they were held in the laboratory at dissolved oxygen concentrations of 4.5 mg/L (Garside 1959).

Trophic linkages

Lake trout are, typically, a top carnivorous species in North American oligotrophic lakes, and their trophic interactions follow the typical pattern of multiple-trophic-level structuring of aquatic ecosystems (i.e. linkages among piscivores, zooplanktivores, zooplankton and phytoplankton).

In boreal forest lakes of northwestern Ontario, the trophic ecology of lake trout and cisco (*Coregonus artedi*) have been determined to differ among lakes with different conductivities (Trippel and Beamish 1993). Heterogeneity in bedrock composition and surficial geology, especially in the boreal forest region, caused regional variation in lake chemistry such as conductivity, total dissolved solids, alkalinity and pH (Brousseau et al. 1985). Since conductivity levels in lakes are known to be positively associated with total phosphorous concentration and the productivity of phytoplankton, zooplankton, zoobenthos and fish (Donald et al. 1980; Chow-Fraser 1991), Trippel and Beamish (1993) speculated that lake-to-lake variability in the degree of impact of top level predators on organisms at lower trophic levels may be a function of inter-lake variation in conductivity. As such, they suggested that conductivity could be used as a surrogate for lake trophic status. In examining the hypolimnetic trophic interactions among lake trout, cisco, zooplankton, and macroinvertebrates in six northwestern Ontario lakes that varied seven-fold in conductivity, Trippel and Beamish (1993) found that faster growth and larger body size of lake trout in high-conductivity lakes were associated with "larger meal sizes," earlier entry of cisco into their diet and consumption of relatively few large-bodied prey. In high-conductivity lakes, cisco were abundant, large bodied and fast growing, and lake trout to cisco ratios were approximately 1:1 or less. In a low conductivity lake (Greenwich Lake, northwestern Ontario), however, this trophic structuring was reversed. In this lake, lake trout were about 12 times as abundant as cisco, and fed on large quantities of age 0 cisco and *Mysis relicta,* a trophic state associated with "top-down" trophic structuring.

Mason et al. (1998) used a size-structured model in the evaluation of the effects of prey fish community dynamics on Lake Superior lake trout, which determined that prey size structure was important for young lake trout but decreased in importance for older lake trout, especially with increasing prey density.

Populations

Lake trout populations are generally characterized by low growth, late maturity, low density, and low annual productivity associated with their oligotrophic environment (Burr 1997). Martin and Olver (1980) have reported that life history traits (such as growth rate and age at maturity) of lake trout are variable and reflect the overall lake productivity and food habits of the population. In Ontario lakes, it has been shown that surface area is related to the complexity of the system and, as such, has a significant effect on the growth, mortality, and productivity of lake trout (Carl et al. 1990; Payne et al. 1990). It has been observed that the numbers of predators and competitors of lake trout as well as the life history traits such as mean size, maximum size, and age at maturity increase with lake size (Carl et al. 1990; Burr 1997).

In relatively small Arctic and sub-Arctic lakes in Alaska, Burr (1997) observed that "the complexity of the fish community, which is strongly related to lake surface area, is an important factor influencing maximum size, maturity schedules, and population density."

A review of lake trout feeding ecology by MacLean et al. (1990) indicated that lake trout at all life stages are omnivorous, opportunistic feeders and that it is the quantity, quality and availability of prey type together with fishing mortality that determine the potential growth patterns of a stock and account for the observed size composition of the population.

Genetic differences, such as hatching and emergence time (Horns 1985), growth rate (Haskell 1952), swimbladder gas retention (Ihssen and Tait 1974), age at maturity, and spawning time (P. Ihssen unpublished data, cited in MacLean et al. 1990) among lake trout stocks have been demonstrated. Ihssen et al. (1988) also showed that sympatric (occurring or overlapping in the same geographical area) lake trout stocks in large lakes do not freely interbreed and are differentiated into reproductively-isolated subunits. Chromosome morphology and banding has also been used to differentiate lake trout stocks (Phillips et al. 1989). There are important management implications of these observed genetic differences among lake trout stocks because these differences are the result of adaptations of populations to local environmental conditions (MacLean et al. 1990).

Critical factors for survival

Generally, lake trout populations require large volumes of cold, well-oxygenated water for survival (Martin and Olver 1980). In the Great Lakes, the optimal habitat boundaries for lake trout have been considered as temperature less than 10 °C and dissolved oxygen (DO) greater than 6 mg/L (Evans et al. 1991). In these large lakes, based on lake trout preferred and physiologically-optimal temperatures, a "fundamental thermal niche" of about 10±2 °C has been suggested (Magnuson et al. 1990). Christie and Regier (1988) have also shown a positive correlation between the "amount of 10±2 °C habitat" in these lakes with lake trout yield. In recent years, however, evidence has been provided that some lake trout populations can also be successful at higher temperatures. A study by Sellers et al. (1998), suggested that the traditional limits of temperature and dissolved oxygen as suggested by Evans et al. (1991) do not adequately describe critical habitat for lake trout, particularly in small lakes. According to the findings of Sellers et al. (1998), the thermal niche of lake trout in small lakes is much broader than previously thought. In boreal lakes surveyed by these authors, lake trout at night occupied epilimnetic waters at 19-20 °C and inhabited oxygenated waters greater than 6 mg/L DO throughout the spring and summer. Thus, the authors concluded that the thermal niche of lake trout in small lakes included the epilimnion. This conclusion is supported by an earlier study (France and Steedman 1996) of lake trout trophic dynamics which provided evidence to suggest that lake trout may be an integral part of the epilimnetic and littoral food web.

In some shallow Ontario lakes, which become isothermal in summer (15 °C to 20 °C), lake trout have been observed to congregate in cool groundwater discharges when ambient temperature exceeds their tolerance range (6 °C to 13 °C) (Snucins and Gunn 1995). These groundwater refugia in relatively shallow lakes need to be protected as critical habitats for lake trout.

Based on information from lakes Simcoe and Manitou (Ontario) and Seneca Lake (New York), Sly and Evans (1996) have provided a synthesis of spawning habitat requirements of lake trout in a dichotomous key "which orders major factors that influence habitat suitability." These requirements are summarized as:

Suitability of substrate size and stability

Gravels of 4-6 cm in diameter are the preferred substrate. In addition the substrate needs to be stable and not subject to ice-push or excessive wave action because developing embryos are sensitive to physical disturbance.

Sediment transport

Successful spawning sites are isolated from bedload transport and are often found near deep waters and on isolated rocky points and islands in small lakes (Edsall et al. 1992).

Water quality

The dissolved oxygen (DO) requirements range from >5mg/L for eyed embryos to >7 mg/L for pre-hatch embryos (Davis 1975). Lake trout embryos held in the laboratory at 4.5 mg/L DO suffered serious developmental abnormalities (Garside 1959). Temperature tolerance limits for embryos are reported to range between 0.5 and 6 °C and pH tolerance is between pH 6.0 and 8.5 (Emerson et al. 1975).

Information deficiencies

In the Arctic, lake trout populations in some lakes migrate into streams in summer for feeding, and return to lakes to over winter (Power 1997). There are also unconfirmed or anecdotal reports of river spawning of lake trout in certain rivers of Alaska (Morrow 1980) and the Back River in Nunavut (Scott and Crossman 1973). However, information is lacking with respect to the extent and nature of use of stream habitat for spawning and feeding by lake trout.

Further work is required to better understand the factors controlling age at maturity (such as availability of preferred food) in lake trout populations.

In temperate North American lakes, optimum substrate size and shape as well as interstitial space have been demonstrated to be critical for site selection by adults and protection and survival of eggs and fry. These factors need to be investigated for lake trout stocks in Arctic lakes to determine if the same holds true in tundra lakes that are subjected to different environmental conditions.

Long-term monitoring over large areas in the Arctic is needed to adequately characterize the optimum water quality requirements for spawning and incubation because of the dynamic nature of biological and meteorological processes that affect them.

Difficulty in sampling early life stages is generally accepted as a reason for the lack of information on early life stages of lake trout (Marsden et al. 1995*b*). Improved sampling methods need to be developed to adequately understand various biological and ecological aspects of early life stages of lake trout.

Lake trout are reported to spawn in both shallow and deep waters. The relative success of spawning at these sites in terms of contribution to year class recruitment should be studied.

Since little information is available on habitat use by young lake trout, behavioral and ecological research is required to determine optimum habitat requirements for this stage.

Some lake trout stocks have been reported to make extensive migrations in the Great Lakes area, while others do not migrate and remain in their natal habitat or a limited part of a lake throughout their life. Since it is possible that lake trout use northern river systems as migratory corridors between lakes, and as feeding and spawning areas, studies on tagged fish are required to determine movement of lake trout in Arctic watersheds.

The intentional removal of spawning habitats produced no detectable change in timing of spawning and fish appeared to rapidly abandon efforts to use the traditional sites. Whether the results of these experiments are applicable in other lakes and areas needs to be explored.

Information on the movement of newly-hatched fry to appropriate rearing and nursery areas is limited.

Lake trout have been reported to congregate in groundwater discharge areas when ambient temperature exceeds their tolerance range (6 °C to 13 °C) (Snucins and Gunn 1995). Since these groundwater refugia could be critical for lake trout survival in both summer and winter, their ecological significance needs to be investigated for lake trout Arctic watersheds.

Since little is known about the predator-prey relationship between lake trout and other fish species in the Arctic, research in this direction is needed to better understand the trophic dynamics in Arctic lakes. It is particularly important in view of the low biodiversity and short food chains in Arctic lakes, which make lake trout, as the top predator, highly vulnerable to changes in trophic structure of Arctic lakes. In particular, the predator-prey relationship between slimy sculpin and age 0 lake trout needs attention because they overlap spatially on reefs and lake trout spawning areas.

Substrate size has been shown to affect predation on lake trout eggs and fry by sculpins. This aspect need to be investigated regarding habitat restoration and compensation.

Table 1. Habitat and biological characteristics of lake trout, *Salvelinus namaycush* (adapted from Ford et al. 1995).

Eggs	
Temperature	0.3-10 °C
Incubation Temperature	5.0 °C
Incubation Time	49-175 days depending upon temperature
Dissolved Oxygen	>6 mg/L; lower lethal level 2.5 mg/L
pH	
Current Velocity	
Substrate	Rubble and cobble
Cover	
Larvae and Juveniles	
Temperature	6-13 °C for juveniles but found in 0.8 °C in Arctic Canada. Prefer 12 °C; upper lethal 23.5 °C
Dissolved Oxygen	Recommended 9.75 mg/L; lower lethal level 3.0 mg/L; 4.5 mg/L retarded growth
Preferred Habitat	Shallow inshore areas of large lakes
Preferred Depth	Variable up to >100 m
Preferred Current Velocity	
Substrate	Cobble/rubble
Cover	Boulders, woody debris
Turbidity/Total Suspended Solids (TSS)	Maximum recommended <10 mg/L TSS
Primary Food	Benthic or planktonic invetebrates
Secondary Food	Fish
Adults	
Temperature	0.0-23.8 °C, optimum for growth 15 °C (no information from Arctic) (Boulva and Simard 1968)
Dissolved Oxygen	Recommended level 7.75 mg/L; the value is a general criterion for freshwater fish calculated by Davis (1975)
Preferred Habitat	Pelagic zone of large lakes
Preferred Depth	Variable
Preferred Current Velocity	
Substrate	No preference
Cover	
Turbidity/TSS	Maximum recommended <10 mg/L TSS
Primary Food	Fish
Secondary Food	Benthic and planktonic invertebrates
Spawning	
Spawning Habitat	Shallow inshore areas of large lakes, occasionally rivers
Spawning Substrate	Preferred substrate rubble/cobble 2-30 cm diameter
Nest Construction	None; broadcast spawners
Spawning Temperature	Preferred temperature 10.0 °C (no information on Arctic (Casselman 1995; Marsden et al. 1995*a*, 1995*b*; Fitzsimons 1995)
Spawning Depth	5-10 m
Spawning Current Velocity	Often chose well-oxygenated areas
Fecundity	1,000-15,000 eggs/female
Age at Maturity	Males 5-13 years of age; females 6-14 years of age

Table 2. Matrix indicating state of information on the biology and ecology of lake trout *(Salvelinus namaycush)* in the Canadian Arctic.

Topics			**Life Cycle Stages**			
			Eggs	Larvae	Juveniles	Adults
Ecology and Habitat Requirements	Critical Habitat	Summer	n/a	**L**	**L**	**M**
		Winter	n/a	**L**	**L**	**L**
	Reproduction	Timing	n/a	n/a	n/a	**H**
		Habitat	**M**	**L**	**L**	**M**
	Rearing and Refugia	Summer	n/a		**L**	**L**
		Winter	n/a		**L**	**L**
	Food		n/a		**L**	**L**
	Water	Depth and Velocity	**L**		**L**	**L**
		Quality	**L**	**L**	**L**	**L**
Behavior	Feeding		n/a			**L**
	Predation/Predator Avoidance		**L**	**L**	**L**	**L**
	Migration		n/a	**L**	**L**	**M**
Physiology/Biochemistry	Growth		n/a	**L**	**L**	**L**
	Function[4]					**M**
Trophic Linkages[1] and Bioenergetics[2]			**L**	**L**	**L**	**L**
Populations[3]			n/a	**L**	**L**	**M**
Human Impacts						**L**

H = High; M = Medium; L = Low; n/a = not applicable. Blank boxes indicate no information available.
1 = Trophic linkages among or between species; 2 = Bioenergetics (balance between energy intake in the form of food and energy utilization by animals for life-sustaining processes) includes role of external factors such as global warming and pollution; 3 = Populations includes limiting factors and vulnerability to environmental changes; 4 = Ability to adapt/tolerate environmental changes.

Table 3. Matrix indicating state of information on the biology and ecology of lake trout *(Salvelinus namaycush)* in areas south of 60°N.

Topics			**Life Cycle Stages**			
			Eggs	Larvae	Juveniles	Adults
Ecology and Habitat Requirements	Critical Habitat	Summer	n/a	**M**	**M**	**M**
		Winter	n/a	**M**	**M**	**M**
	Reproduction	Timing	n/a	n/a	n/a	**H**
		Habitat	**H**	**L**	**M**	**H**
	Rearing and Refugia	Summer	n/a	**L**	**L**	**H**
		Winter	n/a	**L**	**L**	**H**
	Food		n/a	**M**	**M**	**H**
	Water	Depth and Velocity	**M**	**M**	**M**	**H**
		Quality	**M**	**M**	**M**	**H**
Behavior	Feeding		n/a	**L**	**M**	**H**
	Predation/Predator Avoidance		**M**	**L**	**L**	**M**
	Migration		n/a	**L**	**L**	**M**
Physiology/Biochemistry	Growth		n/a	**L**	**L**	**M**
	Function[4]			**L**	**M**	**M**
Trophic Linkages[1] and Bioenergetics[2]			**L**	**L**	**L**	**H**
Populations[3]			n/a	**L**	**M**	**M**
Human Impacts			**L**	**L**	**M**	**H**

H = High; M = Medium; L = Low; n/a = not applicable. Blank boxes indicate no information available.
1 = Trophic linkages among or between species; 2 = Bioenergetics (balance between energy intake in the form of food and energy utilization by animals for life-sustaining processes) includes role of external factors such as global warming and pollution; 3 = Populations includes limiting factors and vulnerability to environmental changes; 4 = Ability to adapt/tolerate environmental changes.

ARCTIC GRAYLING *Thymallus arcticus*

Distribution

According to Scott and Crossman (1973), "Arctic grayling has a holarctic distribution and occurs in northern freshwater drainages, from Hudson Bay west including all of Alaska, St. Lawrence Island, Bering Sea, to the Kara and Ob Rivers of northern Eurasia." Arctic grayling are distributed in lakes and streams throughout the mainland NWT and Nunavut, but have not been reported from the Arctic Archipelago (McPhail and Lindsey 1970; Scott and Crossman 1973). Arctic grayling are one of the most sought-after fish species by anglers and fishermen (Armstrong 1982; Armstrong et al. 1986).

Biology of Life History Stages

Adults: Arctic grayling have two different patterns of life history. Adults of some populations may reside in lakes and utilize inlet or outlet streams for spawning and rearing. Or, the adults may live in rivers or large headwater streams and migrate into smaller headwater tributaries for spawning and rearing (Scott and Crossman 1973; Carl et al. 1977; Northcote 1993, 1995, 1997). In the Canadian north, Arctic grayling are characteristically found in schools in clear water lakes and streams (McPhail and Lindsey 1970).

In the large lakes, they are generally found along rocky shores and near stream mouths (McPhail and Lindsey 1970). The riverine habitat is characteristically cold, clear water with pools and riffles. They are often found downstream of riffles and utilize pools as resting areas but also as food sources (Ford et al. 1995).

Adult grayling are reported to occupy relatively-deep water (up to 1.36 m deep) compared to fry, and current velocities of less than 0.22 m/s (Hubert et al. 1985). During the warmest periods of summer, Arctic grayling inhabit streams and rivers where the average maximum water temperature is between 7.6 and 16 °C and the minimum dissolved oxygen during the summer low-flow period is >6 mg/L (Hubert et al. 1985).

Eggs: The eggs are dense and sticky and become lodged in interstitial spaces of rocky substrates (2-3 cm deep spaces) (Beauchamp 1981, 1982, both cited in Northcote 1995). Most authors have reported that spawning takes place in mid to late afternoon (Northcote 1995). However, in Alaskan streams spawning has been observed in the evening and at night (Van Wyhe 1962). Prior to fertilization, Arctic grayling eggs are generally smaller (2-3 mm) than those of other salmonids, such as Pacific salmon. Fecundity is generally high and may be related to the small size of eggs (Northcote 1995). There is, however, considerable variation in fecundity within a population. For example, in a grayling population near Great Slave Lake (NWT) the fecundity varied from 6,474 to 16,887 eggs per kg body weight (Falk et al. 1982).

McPhail and Lindsey (1970) reported hatching to occur within 16-18 days at 9 °C. However, other authors (Kratt and Smith 1977; Krueger 1981) have reported hatching within 8-32 days at temperatures between 5.8 °C and 15.5 °C.

Larvae and juveniles: The newly-hatched larvae are small and are weak swimmers (Northcote 1995). The size of newly-hatched larvae is reported to be about 14-15 mm in length for the Sukunka River, British Columbia (Stuart and Chislett 1979). In Washington and Montana, they are reported to be even smaller (7-11 mm) at emergence (Watling and Brown 1955; Beauchamp 1981, both cited in Northcote 1995).

Some authors have reported passive downstream movement of grayling fry or alevins in early life stages (Butcher et al. 1981; Kratt 1977). Clark (1992*b*, cited in Northcote 1995) investigating the hypothesis that stream flows during the known periods of egg deposition, hatching and emergence, and larval stage could influence year class strength of grayling in the Chena River (Alaska), found that stream flow was a significant descriptor of variability in recruitment. However, passive drift of fry or alevins would seem to depend on stream flow velocities and needs to be investigated.

Newly-hatched fry are mainly found in side channels of rivers or margins of tributaries in 30-50 cm deep water (Stuart and Chislett 1979; Stewart et al. 1982; Northcote 1995) and gradually move out to deeper and faster waters over boulder, cobble, silt and sand substrates with water current at about 0.8 m/s, away from strong currents (Northcote 1995; Evans et al. 2002). Because of their small size and weak swimming abilities, grayling fry are vulnerable to high stream discharge (Stuart and Chislett 1979). In an Alaskan study (Clark 1992*b*, cited in Northcote 1995) it was found that recruitment during the initial weeks of their life was negatively correlated with stream flow.

Initial schooling of fry has been observed, but they begin to show agonistic behavior within three weeks of hatching. After about six weeks they appear to have developed a hierarchy of small territories (Kratt and Smith 1979). Movement of fry from backwater rearing areas into shallow riffles with silt-free gravel has also been observed (Stuart and Chislett 1979).

Juveniles prefer sand and gravel substrates and use rocks as cover but may also utilize vegetation and shade for this purpose. They concentrate in areas of low current of about 0.8 m/s and shallow depths between 20 and 30 cm (Northcote 1995). According to Ford et al. (1995) "juveniles often overwinter in the same large rivers and lakes as the adults but "in many cases they migrate to the spawning ground areas in the spring 2-4 weeks after the spawners have left." When they are approximately one year old, the juveniles begin their annual migration between natal streams and over-wintering areas (Ford et al. 1995). This sequence is repeated for several years until sexual maturity is reached when adults begin taking their first spawning migration to suitable spawning habitat, often to streams apparently used by their parents (Northcote 1997).

Lake-dwelling grayling are either outlet or inlet spawners and demonstrate intraspecific differences in directional responses to water current (rheotaxis). In laboratory experiments Kaya (1989) demonstrated that in Montana the outlet young had a significantly greater tendency to swim upstream than inlet young, and the young from reciprocal crosses showed intermediate responses. On the basis of these experiments, the author concluded that this intraspecific difference in directional responses to water current appeared to be genetically based but may also be subject to environmental modifications. Similar experimental comparisons between outlet- and inlet-spawning sockeye salmon (*Oncorhynchus nerka,* Brannon 1967, 1972; Raleigh 1971), rainbow trout (*O. mykiss,* Kelso and Northcote 1981; Kelso et al. 1981) and cutthroat trout (*O. clarki,* Raleigh and Chapman 1971; Bowler 1975) have been conducted and similar conclusions have been drawn by these authors.

Kaya (1991) also tested the hypothesis that young Arctic grayling from fluvial populations are adapted to life-long residence in streams by having an innately-greater tendency to hold position and a lesser tendency to go downstream. While the response of 0-10 day old, post swim up grayling tested in an artificial stream contradicted the hypothesis (i.e. they drifted downstream), the responses of the 18-31 day old, post swim up grayling from both fluvial and lacustrine populations were consistent with the hypothesis. Rheotactic differences between the two populations were even more pronounced in trials with 47-72 day old, post swim up grayling.

Ecology of life history stages

Habitat use: Arctic grayling have specific habitat requirements in various seasons and at various life history stages (Northcote 1995). These requirements are critical for survival. Thus, there are migrations that are trophic (feeding), seasonal (over-wintering) or reproductive (spawning).

Spawning: Adults of Arctic grayling have been reported to migrate distances of 60 km to 320 km from their over-wintering refugia in the deep water of rivers and lakes to spawning grounds in inshore, shallow areas (Evans et al. 2002). In the spring, when ice and snow have not quite melted, Arctic grayling leave their over-wintering areas and enter the tributaries of mainstem rivers and lakes (usually at the mouth of tributaries near riffled areas) to spawn (McPhail and Lindsey 1970; Scott and Crossman 1973; Stein et al. 1973; McCart et al. 1976; Machniak and Bond 1979; Ford et al. 1995). They are also reported to spawn in the mainstem rivers (Armstrong 1982). Some lacustrine populations in Alaska are reported to spawn in lakes in shallow waters (0.15 to 0.9 m deep) and over a variety of substrates such as coarse sand, gravel, silt and organic matter (Armstrong 1982). But spawning in lakes usually occurs near outlet and inlet streams in waters less than 1 m deep (Krueger 1981). Spawning of grayling has also been reported in pools as deep as 2.25 m in tributaries to the Liard River (Butcher et al. 1981).

Arctic grayling generally spawn from mid May to early June, but spawning can occur as early as April and as late as July, depending on temperature and latitude (Armstrong et al. 1986). While gravel and rocky bottoms in streams are preferred substrate for spawning, it has been also reported to occur on sandy bottoms (Reed 1964; Scott and Crossman 1973; Beauchamp 1990), and even in highway culverts (Kratt 1981). In the latter case, however, no attempts were made to investigate the success leading to recruitment.

The preferred current velocities in spawning areas are reported to be less than 1.4 m/s. Water temperature at the time of spawning is usually between 7 and 10 ^{o}C but is generally lower in the Arctic (4-5 ^{o}C) (Scott and Crossman 1973; Craig and Poulin 1975; Stewart et al. 1982; Pendray 1983; Beauchamp 1990).

Males set up and guard spawning territories of 6-7 m^2 in riffle areas (Beauchamp 1981, cited in Northcote 1995; Tack 1971). The average water depth over territories established by males is reported to be about 25 cm (Tack 1971). Female Arctic grayling remain in deep pools and enter spawning territories in shallow waters for a short period to spawn (Tack 1971). Spawning behavior of grayling also differs from that of many other salmonids because redds are not constructed or covered by the female.

The post-spawning run into adjoining rivers usually occurs between mid to late June (Chang-Kue and Cameron 1980). Armstrong et al. (1986) have suggested that water temperature and spring flooding are the likely triggers for grayling spawning. In the streams in Alaska, a temperature of about 4 °C appears to trigger grayling spawning (Armstrong et al. 1986) and in the Yukon and northern B.C. the trigger is about 5 °C (Stewart et al. 1982; Pendray 1983).

Rearing: Newly-hatched larvae or alevins spend about 3 to 5 days within the substrate, until absorption of the yolk sac (Krueger 1981; Kratt and Smith 1977; Scott 1985). Newly-emerged fry are usually found in side channels or margins of small natal tributaries where some remain through the summer (Stuart and Chislett 1979). After emergence they have been reported to move out to deeper and faster flowing waters (Stewart et al. 1982). Downstream transport of fry has been suggested by the work of Butcher et al. (1981) in Liard River system. The small size of emerging fry makes them highly vulnerable to high stream discharge (Nelson 1954; Stuart and Chislett 1979). As such, Clark (1992*b*, cited in Northcote 1995) found that grayling recruitment during the initial weeks of life is negatively correlated to stream flow in the Chena River, Alaska. The mechanisms underlying this negative correlation between grayling recruitment during early life history and stream flows are largely unknown and need to be investigated. However, Clark (1992*b*, cited in Northcote 1995) has provided a list of possible consequences of high stream flow after spawning which could influence recruitment and stock size. This list includes the dislodgement of eggs from spawning substrate in shallow waters, downstream drift of grayling (known to be weak swimmers because of poorly-developed fins) into areas of low food abundance, depressed water temperature associated with high stream flow causing slow growth and survival of larvae, and increases in turbidity (often associated with high stream flow) causing egg mortality, low feeding effectiveness and growth rate, and increased susceptibility to predation.

Both mainstem and tributary stream habitats are reported to be used by grayling juveniles and adults in the Parsnip River drainage (Morrow 1980). In general, river grayling populations use backwater sloughs and channels as summer rearing areas but leave them in autumn before freeze up (Craig and Poulin 1975; Hughes 1986).

Work done in Finland on habitat suitability criteria (Nykänen and Huusko 2002) and transferability of habitat preference criteria (Nykänen and Huusko 2004) for larval European grayling (*Thymallus thymallus*) suggests that habitat preference criteria for velocity developed in one river can be successfully used to assess habitat quality and distribution of newly-hatched grayling in another river. The results of this study support the earlier notions of Scott (1985) and Bardonnet et al. (1991) that for newly-emerged larval European grayling, the most limiting physical habitat characteristic appears to be water velocity. Based on their research, Nykänen and Huusko (2004) suggest that "the apparently universal preference for low water velocity (mostly <10 cm/s) is evidently predetermined by the low sustainable swimming speed of larval grayling."

Food supply: Although the diet of YOY Arctic grayling is not well understood according to Jones et al. (2003*c*), most studies suggest that grayling, in general, are opportunistic feeders (Schmidt and O'Brien 1982; Armstrong et al. 1986; Northcote 1995; Jones et al. 2003*c*). An earlier work (Bishop 1967) also revealed that adult grayling in Great Slave Lake (Northwest Territories) were opportunistic feeders and preyed on "all things living or dead." Miller (1946, cited in Scott and Crossman 1973) reported that adult grayling may even consume lemmings in their diet. The fact that grayling at times feed on fish eggs including their own as well as small fish is well documented (Scott and Crossman 1973; Armstrong 1982; Northcote 1995). According to the reviews of Arctic grayling by Armstrong et al. (1986) and Northcote (1995), adult and subadult Arctic grayling appear to primarily feed on drifting insects in streams, but in lakes they are essentially planktivorous. Armstrong et al. (1986) reported that adult grayling forage on benthic invertebrates when invertebrate densities in the water column become low. A survey to measure stable isotopes and dietary composition of Arctic grayling from the Fortymile River watershed (Alaska) showed a high degree of diversity of macroinvertebrates in stomach contents with 9 to 15 taxa of both aquatic and terrestrial forms (Crock et al. 2003).

In lakes, while zooplankton may be a significant part of the diet of grayling, aquatic insects are always a major part of their diet (Armstrong 1982; Armstrong et al. 1986). Lacustrine grayling ranging in size between 5.5 and 8.5 cm tend to locate all lake prey equally on the basis of core body size, with some ability to locate pigmented pond *Daphnia pulex* and pond *Heterocope* at slightly greater distances than other species at high light intensities (Schmidt and O'Brien 1982; O'Brien and Evans 1992). However, these authors observed that grayling failed to attack pigmented copepods (*Diapyomus pribilofensis)*. These copepods were ubiquitous and abundant in the study area (Toolik Lake area, Alaska), are pigmented deep red and are of similar size to *D. pulex*. The authors concluded that grayling were unable to attack these copepods because of their agility and evasiveness to grayling attacks.

In a more recent study (Jones et al. 2003*b*), it was demonstrated that in lake-outlet tundra streams in NWT "food selectivity by YOY grayling was strongly sized-based and YOY primarily consumed larger taxa, especially Chironomidae and Simuliidae" (similar findings were also reported by Birtwell et al. 1984 who examined the diet of YOY and juvenile Arctic grayling in the Yukon). Jones et al. (2003*b*), however, observed that as the YOY grayling grew they captured more large and small prey. These authors concluded that "despite the abundance of lake-derived prey, in-stream production of invertebrates in tundra streams should largely determine the productive capacity of Barrenlands streams" as fish habitat.

Grayling fry are known to begin feeding as early as 4 days after hatching (Brown and Buck 1939, cited in Northcote 1995). Grayling larvae feed mainly on zooplankton, but as they grow bigger, they switch to larval insects, both aquatic and terrestrial (McLeay et al. 1984).

The food preference of juveniles, like the adults, is extremely broad. While their diet primarily includes aquatic insects, they also consume terrestrial insects including bees, wasps, grasshoppers, ants, and beetles. Their diet may also include small fish, fish eggs, and crustaceans (Scott and Crossman 1973). Some authors (Stewart et al. 1982; Elliott

1980) have reported that stomach contents of YOY can contain about 80% benthic organisms. Most YOY (>80%) consume small benthic insects (such as larval chironomids, simuliids and small plecopteran nymphs), but as they grow older they also begin to feed on drifting terrestrial insects (Birtwell et al. 1984; Northcote 1995). In the latter part of summer, the territorial young grayling have been reported to feed mainly on drifting insects in the water column or on the surface. It is important to note that increases in turbidity and suspended solids have been reported to significantly reduce the drift feeding ability of young Arctic grayling. In this regard, suspended solids concentrations of 100 mg/L significantly impaired feeding of young Arctic grayling (McLeay et al. 1987). In laboratory experiments, these authors reported that Arctic grayling were displaced downstream in waters with sediment concentrations ≥100 mg/L. Also, relative to sediment-free control water, at concentrations of 100 mg/L or more, Arctic grayling were slower to consume prey and more frequently missed a food item when they attempted to capture it at the water surface.

The time of grayling feeding activity varies considerably. In general, they appear to feed at or near surface during the day with a mid morning and early-afternoon feeding peak (Northcote 1995). During summer months (long light period) feeding can be continuous but, later in the year they stop feeding at the onset of darkness.

Over-wintering grayling are reported to feed exclusively on benthic organisms (Stuart and Chislett 1979).

Current velocity plays an important role in drift feeding by Arctic grayling. In an experimental stream both the angle and distance at which fish first located each prey were affected by stream velocity (O'Brien and Showalter 1993). Location distance remained unchanged at the comparatively-low velocities (11.6 and 32.3 cm/s) but declined at higher velocities (45.8 cm/s) and, as such, current velocity appeared to compensate for reduced search area. Stream debris which remained suspended in the stream shortened location distance and also decreased location angle. This observation is consistent with the results of experiments conducted by McLeay et al. (1987), which showed that suspended sediment concentrations of ≥100 mg/L can seriously impair feeding behavior of young grayling.

Arctic grayling demonstrate considerable fidelity to feeding grounds in Alaskan waters (Buzby and Deegan 2000; West et al. 1992).

Migration: The migration cycle in stream and river dwelling Arctic grayling populations is complex and involves at least three major types of habitat (trophic, refuge, reproductive). These migrations have been reviewed by Northcote (1993, 1995). However, there is considerable ontogenetic and regional variation in their migratory behavior (Northcote 1997). After hatching in shallow spawning beds, the young fry of both fluvial and lacustrine populations usually undertake a short trophic migration to summer feeding areas in stream margins, stream backwaters or side channels. In autumn, these underyearlings migrate to suitable winter refuges because their feeding areas in natal streams freeze solid or develop frazil ice. This sequence of trophic and refuge migration is repeated for several years until sexual maturity. At this time adults start their first reproductive/spawning migration to suitable spawning areas. Thereafter, the cycle of trophic-refuge-reproductive migrations between feeding, wintering, and spawning habitats in adults is repeated each year for the rest of their life span (Armstrong 1982; Northcote 1995).

Buzby and Deegan's work (2000) is perhaps the most comprehensive conducted on inter-annual fidelity to summer feeding sites of adult Arctic grayling. The study is based on long-term (15-year) records of individually-tagged grayling in the Kuparuk River (Alaska). The Kuparuk River has been the site of a long-term fertilization experiment, which allowed the authors to evaluate the effects of habitat quality on site fidelity. There was no significant difference in fidelity to either the reference or the more-productive fertilized site. More than half of the tagged fish were recaptured within 300 metres of the site where they were captured in previous years. When the numbers of returning fish were adjusted to mortality and capture efficiency of gear (angling), it was estimated that about 74% of the fish returned to the reach. This high return rate was consistent with the 98% return of radio-tagged grayling to the Delta Clearwater River, Alaska (West et al. 1992).

Mark-recapture studies conducted in the Fort Simpson area of the Mackenzie River have provided strong evidence of trophic and reproductive homing (Jessop and Lilley 1975). The study also indicated that Mackenzie River tributaries with open water are important wintering habitat for grayling.

Substantial evidence has been provided to show that during the summer feeding period, Arctic grayling in Alaskan streams adopt a larger-older-fish-upstream distribution pattern (Hughes and Reynolds 1994; Hughes 1998*a*, 1998*b*). However, the mechanisms involved in the maintenance of these population processes are poorly documented (Hughes 1999). Tack (1980) had proposed that most young fish recruit to downstream reaches, and as they grow older and bigger, they move to feeding positions further upstream. The feeding positions held by various ages and sizes were probably caused by competition between old and bigger fish and the younger and smaller ones (Hughes and Reynolds 1994). The work done by Hughes (1999) strongly corroborates this hypothesis and suggests that the size and age gradients of Arctic grayling are the result of a single population distributing itself along the entire length of the river. This distribution pattern in streams adds to the complexity of migrations and habitat selection by Arctic grayling.

Seasonal changes in the habitat use and migrations of the European grayling (*T. thymallus*) in northern Finland have also been documented (Nykänen et al. 2001; Nykänen et al. 2004). They demonstrated that by the end of September, 0.7-1.6 km up- or downstream in the River Kuusinkijoki (66°N) European grayling shifted from riffle sites to deeper and slower pool sites. Based on their results, these authors concluded that: "Adult grayling are seasonally local fish with seasonally differing habitat requirements. The fish tendency to gather into deep pools in autumn suggests that lack of suitable winter habitat is a possible bottleneck to many grayling stocks. In areas with few winter pools, grayling stocks can also become vulnerable to overfishing during the cold season. Maintaining habitat diversity, increasing the amount of over-wintering habitat, setting appropriate fishing restrictions and assuring that fish are able to move between the habitats needed in different seasons should all be considered in grayling conservation" (Nykänen et al. 2004). Nykänen and Huusko (2003) also reported interesting observations regarding size-related changes in habitat selection by larval grayling in the same river. As the larval grayling grew, they shifted from shallow habitats with low water velocities, fine substrata and abundant vegetation cover to deeper sites with higher velocities, coarse substrata and sparse vegetation cover within three weeks. Habitat (water depth and substrata) preference varied between size groups of larvae as listed in Table 4.

Table 4. Habitat features related to the size of larval grayling.

Larval Length (mm)	Depth (cm)	Substrata and Size (mm)	Vegetation Cover (%)	Water Velocity (cm/s)
17-21 (small)	10-30	mud or sand (<2)	10-70	<10
22-25 (medium)	30-90	sand	<40	<10
26-31 (large)	>50	Sand or boulders	<20	10-50

On the basis of these observations the authors concluded that "the strict habitat requirements of the smallest larval group suggest that these habitats, in particular, are important to the early survival of grayling."

Refugia: Over-wintering is probably the most critical period for Arctic grayling. In the Arctic, entire populations of grayling migrate upstream or downstream (depending on the availability of suitable habitats) in certain tributaries to a limited number of refugia (deep pools, river deltas, springs, and lakes) that are less likely to freeze in winter (Armstrong 1982; Northcote 1997, 1995, 1993; Reynolds 1997*a*, 1997*b*). The extent of this migration to over-wintering habitat can be more than 100 km (West et al. 1992). Adult grayling are reported to migrate at rates up to 5 to 6 km/d (Reynolds 1997*a*). Upon reaching a refuge, entire populations or mixed populations may over winter in the limited space of a single spring (Craig and Poulin 1975). It is not clear if over-wintering migration in grayling is "learned" or "genetically programmed." In the former case, stranding and mortality of first-year grayling may be high (Craig 1989). Juvenile grayling in headwater reaches appear to avoid the energetic costs and risks of predation in migrating long distances downstream by moving upstream in a groundwater-fed tributary and spending the winter in pipe-like channels (N. Hughes, pers. comm. quoted in Reynolds 1997*a*). Over-wintering fish undergo physiological and behavioral changes to conserve energy. For example, territoriality demonstrated in summer feeding is replaced by aggregation. Metabolic costs are usually higher in early winter during acclimation but they are lower and more stable over the remainder of the winter (Cunjak and Power 1987; Cunjak 1988).

Fidelity to over-wintering habitat sites in grayling has been suggested by the studies reported by West et al. (1992). Arctic grayling with surgically-implanted radio transmitters were released at their summer feeding sites in three river systems of the Arctic National Wildlife Refuge and were tracked from aircraft to determine patterns of migration to over-wintering refugia. During August-September the tagged fish migrated from the shallow tundra streams into larger streams with migration rates peaking at 5-6 km/d in early September and averaged 1 km/d. The over-wintering sites included deep pools, spring-fed areas and lakes. In view of the limited number of over-wintering sites and the vastness of the study site (North Slope), West et al. (1992) speculated that an individual Arctic grayling returned to the same over-wintering site each year. However, more work is required to establish over-wintering site fidelity for Arctic grayling. Over-wintering refugia need to be considered critical habitat for the survival of Arctic grayling and further work is required with respect to the maintenance of optimum environmental conditions (e.g. dissolved oxygen, turbidity) in these areas.

Behavior

Feeding and predator avoidance: Lake-dwelling Arctic grayling feed on zooplankton until they are about 15 cm in length (Schmidt and O'Brien 1982; O'Brien and Evans 1992), at which point they assume a benthic mode of feeding (O'Brien et al. 1979). However, they will continue to feed on zooplankton if large-sized prey are available (Evans and O'Brien, unpublished data, quoted in Evans and O'Brien 1988). Both a "cruise strategy" i.e. searching for prey while swimming (Schmidt and O'Brien 1982; Evans 1986) and a "saltatory strategy" i.e. obtaining prey while stationary (Evans and O'Brien 1988) have been demonstrated for planktivorous grayling.

O'Brien et al. (2001), studying the functional response of drift-feeding Arctic grayling to changes in prey density and water velocity, showed that feeding rate increased significantly with higher prey densities up to 1.25/L, above which the feeding rate decreased. The current velocities tested in the experiments (25, 32 and 40 cm/s) had no significant effects on feeding rate. Similarly, while search and pursuit time decreased with increasing prey density, neither was affected by current velocities tested.

Physiology and biochemistry

Growth, fitness, survival: The early part of the life cycle is one of extremely rapid growth, up to 118 mm in the first year of life (Kratt and Smith 1979; Liknes and Gould 1987; Kindschi and Barrows 1990). Average growth rates, however, are reported to be about 60-80 mm per year and juveniles grow slower (35 to 45 mm per year) than yearlings (Craig and Poulin 1975), as would be expected from the typically sigmoid growth curves. Some evidence has been provided to suggest that lake populations of grayling may grow faster and larger than river and stream dwelling populations (Machniak et al. 1980; Hubert et al. 1985).

Age composition of Arctic grayling varies between lakes, streams and watersheds. For example, grayling in two Liard River tributaries did not exceed 6+ years whereas those in a Fort Nelson River tributary ranged up to 8+ years in age (Butcher et al. 1981). In Alaska, grayling captured in various systems were between 5 and 12 years in age with lengths ranging between 30 and 40 cm (Hershey et al. 1997; Deegan and Peterson 1992).

Growth rate of grayling also varies from one system to another, especially between northerly and southerly locations (Northcote 1995). For example, in British Columbia grayling from the more-southerly Parsnip tributaries had the highest growth rates, those from Sukunka drainage intermediate (at least from age 4 to 8), and those from the more northerly Adsett Creek (Fort Nelson system) had the lowest growth rate from age 4 to 8 (Butcher et al. 1981).

During summer, the growth of grayling adults and yearlings was reported to be highly variable. In the Kuparuk River (Alaska), on average, adults gained about 40 g between July 1 and August 15. In some years of poorer growth (e.g. associated with low river discharge and high summer temperatures), adult fish even lost weight (Deegan and Peterson 1992). Young of the year, on the other hand, were reported to grow well under these conditions (Hershey et al. 1997). Under high flow and low temperature the metabolic demands of adults are lowered and more prey are available in drift. On the other hand, when the juveniles' food availability exceeds metabolic demands even at low flow, and higher temperature, growth is promoted (Reynolds 1997*a*).

A number of studies conducted in tundra streams (Kuparuk River and Oksrkuyik Creek, Alaska), have demonstrated that growth of both adult and age 0 Arctic grayling is enhanced by nutrient addition i.e. fertilization (Deegan and Peterson 1992; Deegan et al. 1997). However, inter-annual variation in growth can often exceed differences caused by nutrient enhancement (Deegan and Peterson 1992; Deegan et al. 1997; Deegan et al. 1999; Golden and Deegan 1998). That said, however, Deegan et al. (1997) demonstrated that river fertilization increased the growth and condition of YOY and adult Arctic grayling in response to increases in insect size and abundance (Deegan and Peterson 1992). In some years YOY Arctic grayling were twice as large in the fertilized reach versus those in other reaches by the end of the summer feeding period. Also, in these tundra streams, growth of adult and age 0 grayling responded differently to the physical environmental conditions i.e. river discharge and temperature. The work done by Deegan et al. (1999) suggested that while the adult growth appeared to be positively correlated with mean river discharge and not correlated with mean river temperature, the age 0 grayling growth was negatively correlated with discharge and positively correlated with temperature (seemingly an energetics issue related to swimming and to food availability). Similar relationships among growth, river discharge and temperature have been reported for European grayling, *T. thymallus* (Northcote 1995). These observations suggest that river discharge and water temperature may be determinants of the long-term survival and population dynamics of grayling in Arctic tundra streams (Deegan et al. 1999). Furthermore, growth is controlled by an interaction of summer temperature and food availability (Reynolds 1997*a*) and, as such, they are considered critical aspects of grayling habitat; an issue of particular significance in oligotrophic waters (Jones and Tonn 2004).

Deegan et al. (1997) concluded that the density-dependent growth and response to river fertilization by Arctic grayling indicated that these fish were "food limited." Lower food per capita coupled with increased energy demands for aggressive behavior possibly resulted in weight loss in high fish density treatments. In both control and fertilized reaches of the Kuparuk River in Alaska, fish growth, energy storage, and reproductive response of females declined with artificially-increased fish density. Fish growth and energy storage were more closely related to per capita insect availability than algal standing crop.

Buzby and Deegan (2000) examined the return of Arctic grayling to summer feeding areas that bypassed fertilized reaches of the river. In that many fish returned to the same area for summer feeding over a number of years it was deduced that this was an adaptive strategy in cold habitats where the open water period is but a few months. Even though Arctic grayling feed for 24 h during this period of almost continual light (refer to Deegan et al. 1997), it was speculated that such fidelity to feeding areas ensured the over-winter survival of the fish. These authors also considered that it would be energetically inefficient for the fish to search for new feeding areas when those previously used permitted sufficient energy storage to survive the extended winter period of ice cover.

"Using reference streams as standards against which gains and losses of functions (e.g. production of fish) could be quantified," Jones et al. (2003*b*, 2003*c*) and Jones and Tonn (2004) "examined the effectiveness of a 3.4-km artificial stream (Panda Diversion Channel (PDC)) in the Northwest Territories." The stream was, initially, a water diversion channel that facilitated the mining for diamonds in lakes which were eliminated. Once operational, the PDC restored watershed connectivity and provided for fish migration, and spawning and nursery habitat for fish, particularly Arctic grayling. The authors found that "the average mass of YOY grayling at the end of summer was lower (57%) in the artificial stream than in natural streams." The use of a bioenergetics model by the authors indicated that cooler water temperatures in the artificial stream had limited influence on growth. The model suggested that "low amounts of autochthonous and allochthonous organic matter and poor physical habitat in the artificial stream appeared to limit the productivity of benthic invertebrates and fish." Working at the same site, Jones and Tonn (2004) developed resource selection functions (RSFs) for the YOY Arctic grayling in a natural Barrenlands stream and used them to assess the habitat in the artificial stream created as part of a habitat compensation agreement between DFO and BHP Billiton. The application of these RSFs to observed habitat use in the artificial stream indicated an abundance of quality habitat for small grayling, but a paucity for the larger YOY. Based on these observations, it was suggested that "these results reflect an ontogenetic shift in habitat requirements, from simple needs of small YOY to the more complex demands of larger YOY which could not be met by the artificial stream." As such, this inability of the artificial stream contributed to the slow growth of YOYs and the poor productive capacity of the artificial stream. Dion and Hughes (2004), while studying the growth rate of age 0 grayling in Alaskan streams also showed that prey food availability was a limiting factor. Similar conclusions were drawn by Barndt and Kaya (2000) with respect to the failure of non-native Arctic grayling to become established in an intermittent irrigation canal in Montana.

Age at maturity: The age at maturity for Arctic grayling can vary among populations and can range from as young as 2 years in some Montana lakes (Kaya 1990) and the Mackenzie River drainage (McCart et al. 1974) to 5-9 years in some northern Alaska systems (Craig and Poulin 1975; Armstrong 1982; Clark 1992*a*). This regional variation was considered to be related to differences in growth rate and food availability. After attaining sexual maturity Arctic grayling usually spawn each year, but examples exist where this is not so (Scott and Crossman 1973; Falk et al. 1982).

Tolerance and resistance: Grayling fry have been reported to tolerate low levels of dissolved oxygen, and survive levels as low as 1.4 mg/L in winter (Feldmeth and Eriksen 1978, cited in Hubert et al. 1985).

In laboratory tests, grayling underyearlings tolerated short term (4-day) exposure to very high levels (>50,000 mg/L) of fine suspended inorganic and organic sediments (McLeay et al. 1983). Fish survived in turbid waters downstream of a placer mining activity for 4 to 5 days when suspended sediment levels were much lower (McLeay et al. 1983). In a

6-week laboratory study concentrations ≤1,000 mg/L did not affect survival (McLeay et al. 1984). However, it has been observed by others (LaPerriere et al. 1985, cited by Newcombe and Jensen 1996; LaPerriere and Reynolds 1997) that sac-fry were killed at relatively low concentrations of suspended sediment (24-h exposure to 25 mg/L killed 5.7%, and 48-h exposure to 22.5 mg/L killed 14%, of test individuals. Smothering of eggs with sediment has also resulted in their destruction (K. Liber, Director, Toxicology Center, University of Saskatchewan; unpublished data).

Experiments conducted by McLeay et al. (1987) showed that the acute tolerance of warmwater- or coldwater-acclimated grayling to hypoxic conditions was not impaired by suspended solids. In tests with overburden suspensions grayling showed decreased time to death which could be attributed to the sediment's oxygen demand. McLeay et al. (1983) also reported that high concentrations of placer mining "paydirt" increased the time to death as a result of decreased respiratory rate in grayling acclimated in warm water (15 °C).

Median thermal tolerance limit (survival of 50% of test fish for 96 h) of about 24.5 °C for juvenile Arctic grayling from an Alaskan population has been reported by LaPerriere and Carlson (1973). Lohr et al. (1996) determined the critical thermal maximum (CTM) for juvenile Arctic grayling from the fluvial population of the Big Hole River in Montana and found that thermal tolerances increased with acclimation temperature (8.4 °C, 16.0 °C, and 20.0 °C) i.e. 26.4 °C for the 8.4 °C acclimation group, 28.5 °C for the 16.0 °C group, and 29.3 °C for the 20 °C group. The differences in thermal tolerances are likely to be genetically based, for others have determined that populations of more northern stocks of the same species of salmon have lower tolerances to elevated temperature (Beacham and Withler 1991).

Impacts of human interventions: Northcote (1995) has provided an overview of the effects of human interventions on Arctic grayling and its habitat. The human activities that are known to impact Arctic grayling include impoundments and hydroelectric dams (Lindsey 1957; Withler 1959; Barrett and Halsey 1985, cited in Northcote 1993; Weagle and Cameron 1974; Vincent 1962, cited in Northcote 1995); oil and gas pipelines (McCart 1974, cited in Northcote 1995, Moles et al. 1979); mining (McLeay et al. 1983, 1984, 1987; Birtwell et al. 1984; Lloyd et al. 1987, cited in Northcote 1993; Reynolds et al. 1989, cited in Northcote 1993); roads and culverts (Jones et al. 1974, cited in Northcote 1993); pollution (Wojcik 1955; LaPerriere and Carlson 1973; Peckham 1984, cited in Northcote 1984). All these activities are reported to have impacted Arctic grayling populations either directly or indirectly (Northcote 1995).

Impacts of suspended sediment as a result of placer mining on Arctic grayling are well documented. Damage to gills, dietary deficiencies and slower maturation of grayling held in streams carrying placer mining sediment were demonstrated by Simmons (1984). Evidence of severe damage to grayling populations as a result of the loss of spring and summer habitat due to placer mining (Weber and Post 1985; Lloyd et al. 1987 cited in Northcote 1993; Reynolds et al. 1989, cited in Northcote 1993) and decline in sport fishing use of Alaskan streams was reported by Townsend (1987, cited in Northcote 1993).

Loss of spring and summer habitat for reproduction has also been reported (Weber and Post 1985; Lloyd et al. 1987, cited in Northcote 1993; Reynolds et al. 1989, cited in Northcote 1993;). McLeay et al. (1987) in laboratory experiments showed that suspended sediment in concentrations of 1,000 mg/L or less under otherwise optimal environmental conditions did not cause direct mortality of grayling. However, sublethal effects including impaired growth, decreased scope for activity and decreased resistance to other environmental stressors occurred at placer mining suspended sediment concentrations of ≥100 mg/L. At 300 mg/L suspended sediment was shown to cause increased metabolic rate (e.g. increased oxygen consumption) by McLeay et al. (1984). These authors also observed the downstream displacement of grayling caused by sediment concentrations above control water values (minimum treatment value was 100 mg/L).

Infrastructure development for mining sites such as road construction and culverts have been reported to have detrimental effects on grayling and other salmonids. Road construction has the potential to cause high sediment loads in streams (Rosenberg and Snow 1975). Similarly, stream culverts at road crossings may be velocity barriers for small fish (Stein et al. 1973).

Trophic linkages

A few studies have been conducted on competition for food with other species of fish. Studying the food of grayling, whitefish, and lake charr in Tangle Lakes (Tanana River drainage, Alaska), Wojcik (1954, cited in Armstrong et al. 1986) concluded that these three species did not compete with each other for food. In contrast, Pearse (1974, cited in Armstrong et al. 1986) found that Arctic grayling and round whitefish preferred and possibly competed for the same food items. Similarly, Craig and Poulin (1975) reported that when Arctic charr and Arctic grayling occur together, they may feed on similar food items.

Arctic grayling have several potential fish competitors. Northcote (1993) identified at least 11 of them which included lake whitefish, longnose sucker, and slimy sculpin in waters tributary to Williston Reservoir in British Columbia. In this watershed, mountain whitefish (*Prosopium williamsoni*) are more abundant than grayling. But despite the abundance of mountain whitefish which are earlier spawners, the grayling underyearlings outgrew underyearling mountain whitefish in the Sukunka system (Stuart and Chislett 1979), suggesting that grayling may be a stronger food competitor.

In the tributaries of Williston Reservoir (British Columbia), Northcote (1993) also identified four fish species (bull trout, northern squawfish, northern pike, and burbot) as potential predators of Arctic grayling. Other potential competitors of grayling could be slimy sculpin (*Cottus cognatus*) and suckers (*Catostomus* sp.).

Golden and Deegan (1998) provided experimental evidence to suggest that "young (0+) Arctic grayling have the potential to control the trophic structure of Arctic tundra streams through consumption, nutrient excretion, and the modification of prey behavior." They investigated the effect of young grayling on three trophic levels (algae, invertebrates, and fish) by manipulating fish density and by fertilizing the river (Kuparuk River, Alaska) with phosphorus. They found that "epilithic chlorophyll *a* increased with increased fish density in the fertilized zone only, and the final mean mass of young grayling in the 40 fish·m^{-2} cages was lower than mean mass in the 4 fish·m^{-2} cages." They, therefore, concluded that "young grayling may produce a top-down cascading trophic effect in areas where nutrients are not limited," and "river nutrient status and river discharge may modify the strength of top-down control by young grayling."

Populations

Little is known about the available food supply for various stocks at critical life history stages. The available information suggests that habitat and food availability are highly variable for Arctic grayling. Northcote (1993), reviewing the literature on the size of grayling populations, found that the abundance of adult grayling in rivers can range over three orders of magnitude (low 10s to >1,000 per kilometre).

As noted earlier, grayling use different habitats (spawning habitat, underyearling feeding habitat, juvenile, subadult and adult wintering habitats, and subadult and adult feeding habitats) during their life.

The series of habitats utilized by grayling throughout their life appear to be highly stock specific. Information has been generated that suggests multiple origins for this species in at least two glacial refugia, the Yukon basin and North Slope (Arctic) drainages (McCart and Pepper 1971). A series of studies on Arctic grayling in Alaska have found that there are distinct differences in spawning sites, feeding sites, growth rates and migratory behavior among stocks (Northcote 1995). But the migration and mingling of these stocks from one watershed to another has also been reported (Tack 1980). Phenotypic differences between these stocks have also been reported (Hop 1985; Hop and Gharrett 1989, quoted in Northcote 1995). In northeastern Russia, complex intraspecific diversity among Arctic grayling (*T. arcticus*) has been reported and groups have been separated at the subspecies level. Two groups have been identified on the basis of dorsal fin markings which correspond to the subspecies *T. arcticus pallasi* and *T. arcticus mertensi* (Kurlykova and Makoyedov 1995).

Recent studies (Haugen 2000; Haugen and Vollestad 2000; Koskinen 2002; Koskinen et al. 2000; Koskinen et al. 2002) on European grayling using biochemical and molecular techniques have identified genetic lineages and postglacial colonization of European grayling (*Thymallus thymallus*). For example, Haugen and Vollestad (2000) have clearly demonstrated population differences in early life history traits which suggest that natural selection has resulted in local adaptation in a time period of 13 to 18 generations.

The population size and density of Arctic grayling have been reported to vary considerably and the differences can be attributable to habitat suitability as well as human interventions (Northcote 1995). DeCicco et al. (1997) studied the characteristics of a lightly-exploited population of Arctic grayling in the Sinuk River (Seward Peninsula, Alaska). On the basis of growth, age structure, and abundance (individuals attained large maximum size, lived long and had low annual mortality), it was concluded that this population in the Sinuk River was "static" and experienced low recruitment.

Critical factors for survival

Since Arctic grayling utilize a series of habitats during their life, appropriate food availability at these habitats could be critical for grayling survival. As noted previously, the variable abundance of grayling adults (low 10s to >1,000 per kilometre of river) suggests that habitat and food availability are also highly variable. The factors causing this variability could be critical for grayling survival especially in oligotrophic Arctic and sub-Arctic areas (Jones and Tonn 2004). Consequently the availability and maintenance of optimal environmental conditions in all habitats and especially over-wintering refugia are likely critical to survival.

The habitat and biological characteristics of Arctic grayling are summarized in Table 5. Tables 6 and 7 provide a qualitative assessment of the state of knowledge about the biology of Arctic grayling in the Arctic and areas south of 60°N respectively.

Information deficiencies

A considerable body of information is available on Arctic grayling from Alaska, but not on those from the Canadian north. In particular, information is required on optimal habitat conditions in the Arctic, species associations and predator-prey relationships, fidelity to over-wintering, spawning, rearing, and feeding areas; over-wintering refugia and habitat conditions under ice.

Table 5. Habitat and biological characteristics of Arctic grayling, *Thymallus arcticus* (adapted from Ford et al. 1995).

Eggs	
Temperature	2-16 °C
Incubation Temperature	Optimum 6-10 °C
Incubation Time	8-32 days; 16-18 days at 9 °C
Dissolved Oxygen	
pH	
Current Velocity	Recommended <0.3 m/s
Substrate	Gravel, boulder (<20% sand)
Cover	1-3 cm within gravel
Larvae and Juveniles	
Temperature	2-24.5 °C; optimum for growth 1-12 °C
Dissolved Oxygen	Lower lethal level 1.4 mg/L
Habitat	Streams
Preferred Depth	<50 cm; newly hatched in side channels 30-50 cm deep
Preferred Current Velocity	<0.5 m/s; for European grayling, *T. thymallus* mostly <10 cm/s) Nykänen and Huusko (2004)
Substrate	Gravel, cobble, sand
Cover	Boulders
Turbidity/TSS	<50 mg/L TSS
Primary Food	Insect larvae
Secondary Food	Variable
Adults	
Temperature	1-20 °C, optimum for growth 10 °C
Dissolved Oxygen	Prefer >6 mg/L; lower lethal level 2.0 mg/L
Habitat	River, lake, stream
Preferred Depth	<10 m common, typically 1-2 m depth
Preferred Current Velocity	0.2-0.8 m/s
Substrate	Gravel, rocks, boulders
Cover	Boulders
Turbidity/TSS	Adverse effects at ≥100 mg/L TSS
Primary Food	Aquatic and terrestrial insects
Secondary Food	Variable
Spawning	
Spawning Habitat	Streams
Spawning Substrate	Gravel with <20% sand
Nest Construction	Broadcast spawners but rudimentary redds observed
Spawning Temperature	7-10 °C
Spawning Depth	Variable
Spawning Current Velocity	0.3-1.5 m/s
Fecundity	3,000-15,000; average 4,000-7,000 eggs/female
Age at Maturity	3-9 years

Table 6. Matrix indicating state of information on the biology and ecology of Arctic grayling *(Thymallus arcticus)* in the Canadian Arctic.

Topics			**Life Cycle Stages**			
			Eggs	Larvae	Juveniles	Adults
Ecology and Habitat Requirements	Critical Habitat	Summer	n/a	**M**	**M**	**H**
		Winter	n/a			**L**
	Reproduction	Timing	n/a	n/a	n/a	**H**
		Habitat	n/a	n/a	n/a	**H**
	Rearing and Refugia	Summer	n/a	**M**	**M**	**M**
		Winter	n/a	**L**	**L**	**M**
	Food		n/a	**L**	**L**	**H**
	Water	Depth and Velocity	**L**	**L**	**L**	**L**
		Quality	**L**	**L**	**L**	**L**
Behavior	Feeding		n/a	**L**	**M**	**H**
	Predation and Predator Avoidance			**L**	**L**	**M**
	Migration		n/a	**L**	**L**	**H**
Physiology/Biochemistry	Growth		n/a	**M**	**M**	**M**
	Function[4]		**L**	**L**	**L**	**L**
Trophic Linkages[1] and Bioenergetics[2]			**L**	**L**	**L**	**H**
Populations[3]			n/a			**M**
Human Impacts						**L**

H = High; M = Medium; L = Low; n/a = not applicable. Blank boxes indicate no information available.
1 = Trophic linkages among or between species; 2 = Bioenergetics (balance between energy intake in the form of food and energy utilization by animals for life-sustaining processes) includes role of external factors such as global warming and pollution; 3 = Populations includes limiting factors and vulnerability to environmental changes; 4 = Ability to adapt/tolerate environmental changes.

Table 7. Matrix indicating state of information on the biology and ecology of Arctic grayling (*Thymallus arcticus*) in areas south of 60°N.

Topics			Life Cycle Stages			
			Eggs	Larvae	Juveniles	Adults
Ecology and Habitat Requirements	Critical Habitat	Summer	n/a	L	M	H
		Winter	n/a			L
	Reproduction	Timing	n/a	n/a	n/a	H
		Habitat	n/a	n/a	n/a	H
	Rearing and Refugia	Summer	n/a	L	L	H
		Winter	n/a			L
	Food		n/a	L	L	H
	Water	Depth and Velocity	L	L	L	L
		Quality	L	L	L	L
Behavior	Feeding		n/a	L	L	H
	Predation and Predator Avoidance		L			L
	Migration		n/a	M	M	H
Physiology/Bio-chemistry	Growth		n/a			L
	Function[4]		L	L	L	L
Trophic Linkages[1] and Bioenergetics[2]			L	L	L	L
Populations[3]			n/a			M
Human Impacts						L

H = High; M = Medium; L = Low; n/a = not applicable. Blank boxes indicate no information available.
1 = Trophic linkages among or between species; 2 =Bioenergetics (balance between energy intake in the form of food and energy utilization by animals for life-sustaining processes) includes role of external factors such as global warming and pollution; 3 = Populations includes limiting factors and vulnerability to environmental changes; 4 = Ability to adapt/tolerate environmental changes

LAKE WHITEFISH *(Coregonus clupeaformis)*

The habitat and biological characteristics of lake whitefish are summarized in Table 8. Tables 9 and 10 provide a qualitative assessment of the state of knowledge about the biology of lake whitefish in the Arctic and areas south of 60°N respectively.

Distribution

In Canada, lake whitefish occur from the Atlantic coastal watersheds, west through Quebec (including Ungava Bay), Ontario (including the Great Lakes), Manitoba (including coastal drainages in Hudson Bay), Saskatchewan (mostly northern waters), Alberta and British Columbia (Scott and Crossman 1973). The species is distributed throughout NWT and Nunavut.

Lake whitefish (*C. clupeaformis*) present considerable zoogeographic and taxonomic complexities. As such, Lindsey et al. (1970) concluded that the lake whitefish (*C. clupeaformis*) is composed of a complex of closely-related sympatric species, including the Alaskan whitefish (*C. nelsonii*) and the humpback whitefish (*C. pidschian*). The taxonomic characteristics such as gill raker count, size, and shape in this complex are highly variable depending on geographical area, latitude, and variations in life history (Lindsey et al. 1970). In fact, some reviews of lake whitefish (Richardson et al. 2001; Evans et al. 2002) have included all these three species as one topic.

Lake whitefish occur mainly in lakes but can be found in larger rivers and brackish waters and, as such, exhibit lacustrine, adfluvial and anadromous life history forms (McPhail and Lindsey 1970; Scott and Crossman 1973). It has been suggested that these life history forms and related associations with particular types of water bodies may correspond with particular taxa that make up the whitefish complex (Evans et al. 2002).

Lake whitefish have not been reported from saline waters (Evans et al. 2002). They appear to be less tolerant of saline waters than most other coregonine species (Reist and Bond 1988).

Biology of life history stages

Adults: Adults are closely associated with the benthos in cool, well-oxygenated lakes and are benthic feeders over most of their range (McPhail and Lindsey 1970; Scott and Crossman 1973). Preferred depth is reported to be between 10 and 100 m or deeper. They have been reported to have no specific preference for types of substrate except, at the time of spawning, when they use shallow riffle areas in rivers or shallow beaches in lakes with gravel, rock or sand (Qadri 1968; Scott and Crossman 1973).

Diet varies among fish from different lakes depending on prey availability, which includes aquatic insect larvae, molluscs, and amphipods. Amphipods are the most common food item. Lake whitefish are also reported to prey on fish and fish eggs (Scott and Crossman 1973). In the absence of benthic food resources, planktonic organisms are important food for adults. Gill raker morphology is reported to play a role in diet selection. Populations with larger numbers of short gill rakers consume a greater proportion of benthic organisms than do those with long but fewer gill rakers (Kliewer 1970).

Larvae and juveniles: At hatching, length varies between 10 and 18 mm and larvae grow rapidly throughout the summer months (Morrow 1980). Hatched fry either move downstream in smaller streams or remain in shallow inshore areas adjacent to spawning grounds for about two months. With the rise in temperature, they move into deeper waters, where they gradually adopt a benthic mode of life.

Small changes in zooplankton density and larval abundance have been shown to significantly influence growth rate of juveniles in Lake Michigan (Freeburg et al. 1990). They can tolerate temperatures between 12 °C and 20 °C, but prefer temperatures around 14 °C (Reckhan 1970; Hoagman 1973*a*, 1973*b*).

Eggs: Eggs are adhesive and are broadcast over rocky, sandy or gravelly substrates. No parental care, by either male or female, has been reported. Eggs settle in rocky crevices where they are fertilized by males. Egg survival depends on environmental factors such as current velocity, wind-driven wave action and presence or absence of predators (McPhail and Lindsey 1970; Scott and Crossman 1973).

Ecology of Life History Stages

Habitat use: Lake whitefish are mostly lacustrine in habit, but adfluvial, riverine and anadromous forms have been reported (Scott and Crossman 1973). Although they are essentially benthic, they may be found in the pelagic zones of lakes where they feed on planktonic organisms (Scott and Crossman 1973; Morrow 1980). Adults prefer cooler-temperature waters but are reported to have no preference for substrate (Qadri 1968) until spawning when sandy, rocky or gravelly substrates are preferred.

Spawning: Spawning occurs in lakes, rivers and streams. Unlike many other species, flowing water is not required for spawning (Bryan and Kato 1975). In Canada, anadromous lake whitefish are known to spawn in the Mackenzie River and its tributaries in September and October at temperatures between 1 °C and 1.5 °C (Jessop and Lilley 1975). Following hatching, fry move downstream with spring runoff to the Mackenzie Delta by May or June where they rear in delta lakes and channels and remain associated with the delta until they are about four years old (refer to Evans et al. 2002). Upstream spawning runs have been reported in the Slave River (NWT) in October (McLeod et al. 1985, cited in Evans et al. 2002), Little Buffalo River, Great Bear River (McCart 1982, cited in Evans et al. 2002), and Peace River (Patalas 1993, cited in Evans et al. 2002).

Spawning typically occurs in October, November, and December but the season can extend from mid September to late January throughout the geographic range of the species, likely dependent on environmental factors such as temperature and photoperiod (Lindsey et al. 1970; Scott and Crossman 1973). The water temperature at spawning is usually about 6 °C or less. Laboratory studies conducted by Bernatchez and Dodson (1985) suggested that "the combined effect of high water velocities and low ambient temperature on coregonines' metabolism and swimming performance may be a more important factor than specific spawning temperature in the timing of the early reproductive migration of anadromous coregonines…."

Some authors (Lindstrom 1970; Machniak 1975*b*) have reported that river spawning generally occurs earlier than does lake spawning. In the Great Slave Lake system (NWT), however, both occur at the same time (Bradbury et al. 1999).

In northern latitudes (Arctic and sub-Arctic), spawning may be intermittent depending on trophic conditions and the length of the growing season. But in more southerly lakes whitefish spawn each year (Johnson 1976).

Average fecundity is about 50,000 eggs per female (range 6,000 to 150,000) (Morrow 1980). Healey and Nicol (1975) found no significant differences in either slope or intercept for the regression of $\log_e$ fecundity on $\log_e$ fork length among samples of whitefish from four lakes near Yellowknife (NWT). Fish from relatively southern locations (Buck Lake, Alberta and Lake Erie) had high relative fecundities and fish from Great Slave Lake had low relative fecundity. These differences in fecundity among populations may be related to a variety of factors, including nutritional state, time of sampling, maturity stage and racial characteristics (Healey and Nicol 1975).

Spawning occurs at night and preferred spawning habitat is shallow riffle areas in rivers or shallow beaches in lakes with gravel, sand or rock substrate (Morrow 1980). Depth of spawning areas varies between 1 and 3 m (Morrow 1980). Females randomly broadcast adhesive eggs over the substrate where they settle in rocky crevices and are fertilized by waiting males (Hart 1930, cited in Ford et al. 1995). Eggs incubate over winter for 20 to 23 weeks at temperatures around 0.5 °C (no eggs survived at 0 °C (Morrow 1980)). The work done by Tait (1973, cited in Ford et al. 1995) and Brooke (1975), however, determined that the optimum egg incubation temperature range was between 4 and 6 °C, which conferred greater survival rates than incubation at higher temperatures. Brooke (1975) also found that the time from fertilization to median hatch was inversely related to temperature and ranged from 42 days at 10 °C to 182 days at 0.5 °C. The 1975 laboratory work by Brooke also demonstrated that hatching success was significantly reduced at higher (10 °C) and lower temperatures (0.5 and 2 °C), the number of abnormalities was lowest at intermediate temperatures (4 to 7.8 °C), and eggs incubated at higher temperatures produced smaller post-hatch larvae. A concentration of ≥ 8 mg/L dissolved oxygen is necessary for normal development from fertilized to late eyed-egg stage of lake whitefish (Fleuchter 1980).

Settling of suspended particulate matter in spawning areas adversely affects egg survival (Fudge and Bodaly 1984) and high turbidity can also interfere with the ability of newly-emergent fry to capture food (Hartman 1973, cited in Ford et al. 1995).

Eggs usually hatch in April or May (Scott and Crossman 1973; Freeberg et al. 1990). The role of ice cover on the survival of eggs is not clear as there is contradictory information in the literature. For example, Miller (1956) reported significant mortality in lake whitefish, but Freeburg et al. (1990) reported a nine-fold increase in egg survival under ice cover in two subsequent years. These reported differences emphasize the need for further investigation under Arctic conditions.

Adults of lacustrine populations leave spawning grounds shortly after spawning in the fall and return to deeper over-wintering waters (Ford et al. 1995).

Rearing: Hatched fry (1.1 to 1.4 cm long) grow rapidly during the summer but growth slows abruptly in September. By the end of October larvae are about 12 cm in length (Morrow 1980). Hatched fry either move downstream in smaller streams or *remain in shallow inshore lake areas adjacent to spawning grounds for* about two months. These habitats are, therefore, critical for survival (recruitment) and their significance under Arctic conditions needs to be investigated.

With the rise in temperature in epilimnetic waters, juveniles move into deeper metalimnetic waters, thereby gradually assuming a benthic feeding mode of life typical of adults.

Yolk sac is absorbed within three weeks and fry start feeding on small zooplankters <2 mm in length (Teska and Behmer 1981). Freeburg et al. (1990) demonstrated that small changes in zooplankton density and larval abundance have significant influence on growth rate of lake whitefish in Lake Michigan. As such, availability of appropriate-size food is critical for growth and survival.

Larvae can tolerate temperatures between 12 °C and 20 °C, but prefer temperatures around 14 °C (Reckhan 1970; Hoagman 1973*a*, 1973*b*). Young of the year are generally found over gravel, cobble or boulder substrate and typically remain in these habitats for 6-8 weeks (Faber 1970).

Food supply: According to Morrow (1980), "food of lake whitefish varies with size and age of the fish, location, and the type of food available." During the first year of life, the food habits of lake whitefish change from planktonic to benthic. Larvae and juvenile fish feed on plankton and aquatic insect larvae (Freeberg et al. 1990). In Lake Huron, initial food of young lake whitefish consists of copepods and cladocerans, and by early summer they begin to consume benthic organisms (Reckhan 1970). Adult lake whitefish are essentially benthic feeders. They are reported to feed on snails, clams, chironomid larvae and small fishes (McPhail and Lindsey 1970; Scott and Crossman 1973).

Migration: According to Morrow (1980), movement of lake whitefish in large lakes in general consists of four stages. These are travel from deep to shallow water in the spring, movement back into deep water during the summer as shoal water warms, migration back to the shallow-water spawning areas in the fall and early winter, and post-spawning movement back to deeper water.

Spawning migrations of lake-dwelling populations typically occur in the fall months depending on latitude, earlier in more northerly locations (MacKay 1963, cited in Ford et al. 1995; Scott and Crossman 1973; Morrow 1980). Spawning migration involves moving into shallow areas of lakes or ascending rivers when water temperature cools to between 4.5 and 10 °C (Morrow 1980).

In a study utilizing integrated acoustic telemetry (IAT) to quantitatively express spawning movement and substrate selection during two consecutive spawning seasons (1995 and 1996) in a small boreal lake in Ontario, Anras et al. (1999*a*, 1999*b*) observed high fidelity of lake whitefish toward specific substrata (gravel boulders, etc.) and slope characteristics. These authors observed that "lake whitefish returned to the same locations in both study years [1995 and 1996], but a greater dispersal of spawning locations occurred in 1996 coincident with an increase in nearshore substratum availability of 17% because of water level manipulation." Site fidelity of lake whitefish to spawning grounds requires investigation in the Arctic.

During spawning migrations, lake whitefish have been recorded to move close to the bottom and take little food (Qadri 1968). In some populations males and females move in separate schools, males arriving earlier at spawning locations than females (Machniak 1975*b*). In the Mackenzie River, adults migrate downstream during late fall to over-winter in the delta (Evans et al. 2002).

Refugia: Deep waters in lakes and rivers are considered to be over-wintering refugia due to the post-spawning runs in the fall to these areas (Scott and Crossman 1973).

Behavior

Feeding and predator avoidance: The food habits of lake whitefish during the first year of life change from planktonic to benthic. As plankton feeders, larval lake whitefish compete with other planktivores such as lake herring. In a mesocosm study Todd and Davis (1995) provided information that implied "that diet similarities of juvenile lake herring and lake whitefish in addition to the larger size and more aggressive behaviour of larval lake whitefish resulted in the depressed growth and poorer survival of lake herring." The transition from a planktonic to a benthic mode of feeding is gradual and apparently learning, local food availability, and opportunism play important roles in this transition (Reckhan 1970; Ford et al. 1995) Benthic food includes dipteran larvae, ostracods, gastropods, amphipods, isopods, pelecypods and fish eggs (Reckhan 1970).

Since eggs are broadcast they are highly vulnerable to predation by other fish. For example, Nester and Poe (1984) reported intense predation by longnose suckers on lake whitefish eggs in Lake Huron. It is believed that substrate size can affect the egg losses to predators. Cobble and rubble, which provide interstitial spaces, would likely reduce egg predation (Nester and Poe 1984).

Both burbot and longnose suckers are reported to prey on lake whitefish eggs (Green and Derksen 1987).

Physiology and biochemistry

Growth, fitness and survival: anadromous populations grow slower than lacustrine populations (Lawrence et al. 1984, cited in Evans et al. 2002). The fish mature between ages 7 and 13, and can live to age 20 to 34 (refer to Evans et al. 2002). Based on a review of studies done on lake whitefish populations, Ford et al. (1995) identified three general population types that differed in their growth rates and size at maturity. Heavily-exploited populations from large lakes are characterized by high asymptotic size and size at maturity, but low age at maturity (Healey 1975). Unexploited populations in northern lakes are characterized by low asymptotic size and high age at maturity (Edsall 1960; Johnson 1976). Exploited populations in small lakes are characterized by early age at maturity and small asymptotic size (Johnson 1976). Increased growth in many lake whitefish populations has been attributed to exploitation (Healey 1975; Jensen 1981). This may be a partial compensatory mechanism in response to exploitation and other environmental factors that affect annual mortality (Ford et al. 1995).

Bidgood (1973) observed the divergence in growth patterns of lake whitefish in two similar Alberta lakes (Pigeon Lake and Buck Lake) and attributed the declining growth rate in Pigeon Lake over a period of 12 months to the reduction of predators by the removal of habitat and increased angling pressure which accelerated recruitment to both the whitefish population and those of other competing benthic-feeding forage fish. This aspect of predator-prey relationships needs to be investigated for fish in lakes and streams.

In laboratory experiments, Taylor and Freeberg (1984) noted that larval whitefish survived on low food rations for about 2 weeks without suffering any mortality. These authors suggest that this could be an adaptive response since these fish hatch in late winter to early spring when zooplankton densities, and hence potential food, are generally low. These authors also note that larval mortality would be severe in the absence of any improvement in zooplankton abundance by 2-3 weeks time after hatching.

In laboratory experiments with eggs and larvae of two stocks of Lake Michigan lake whitefish, Brown and Taylor (1992) found that “the length at hatching of larval whitefish was primarily a function of egg size as measured by the overall egg caloric content, and endogenous growth of larvae was highly dependent on the egg lipid content and mg lipid per egg.” These authors also found that “exogenous growth and survival of larval lake whitefish were positively related to prey resource availability and both were sensitive to small changes in the availability of zooplankton prey resources.”

Based on the results of these studies, therefore, it appears that variation in egg composition has the potential to be important in determining larval growth, survival, and eventual recruitment of lake whitefish. In addition, zooplankton abundance and availability could be a critical factor in establishing year class strength in whitefish populations.

Age at maturity: Several factors, including latitude, productivity of the water body and the degree of exploitation appear to determine the age and size at maturity (Healey 1975). In general, lake whitefish in isolated and unexploited populations do not mature until age 6 (Ford et al. 1995). In exploited populations, however, earliest documented maturity is age 2 for males and age 3 for females (several authors cited in Ford et al. 1995). In northern locations such as Great Slave Lake both sexes of lake whitefish may reach sexual maturity at age 13 (Kennedy 1953).

Tolerance and resistance: In a laboratory study, Edsall (1999*a*, 1999*b*) found that juveniles grew heaviest and longest at 18.1 °C and concluded that the “fundamental thermal niche” for juvenile lake whitefish is 15.5 °C to 19.5 °C, which is similar to that reported for other Great Lakes fish such as the bloater (Edsall and Frank 1997).

Little information is available on the limits of hypoxic tolerance by lake whitefish. However, it has been reported that the deteriorated dissolved oxygen conditions in the hypolimnion of Lake Erie caused abandonment of this area by lake whitefish as a summer refuge and their movement to areas where levels were higher (Kenyon 1978). Qadri (1968) also hypothesized that hypoxic conditions may limit lake whitefish movement in summer months in lakes with thermal stratification.

In laboratory experiments, lake whitefish avoided Cu, Pb, and Zn at or below concentrations considered safe (2 μg/L for Cu and Pb, 30 μg/L for Zn as per the Canadian Water Quality Guidelines) without a competing factor such as shade (Scherer and McNicol 1998). When shade was provided the fish did not avoid these metal concentrations, thus increasing the potential for exposure.

Impacts of human interventions: In a field study in Southern Indian Lake, Manitoba, Fudge and Bodaly (1984) stated that the survival of lake whitefish eggs was significantly higher for those incubated in cages designed to minimize exposure to sedimentation, compared with survival in cages allowing full exposure to sedimentation (0.03 to 0.14 g dry wt. sediment/cm^2; deposition of 1-4 mm in depth). The study concluded that "winter sedimentation rates and whitefish egg survival were negatively correlated for cages designed to minimize exposure to sedimentation, while egg survival in the exposed cages was uniformly low."

Sewage discharges can have a detrimental effect on lake whitefish, especially the emergent fry and YOY because they are not only sensitive to sewage they are also known to be attracted to it (Alexander et al. 1977).

In laboratory experiments it was demonstrated that "when whitefish encounter competing gradients of Cd and light, attraction to favourable light conditions can strongly suppress avoidance of toxic concentrations of Cd even over an extended period" (McNicol et al. 1999).

Trophic linkages

Lake whitefish share habitat with a number of other predatory fish such as lake trout and burbot. As such, they not only compete for food with them, their eggs and young are also vulnerable to predation by other fish. Lake whitefish eggs are particularly vulnerable to predation by other fish since they are broadcast and are afforded no parental care. For example, Nester and Poe (1984) reported intense predation by longnose suckers on lake whitefish eggs in Lake Huron. These authors, however, pointed out that an appropriate substrate (cobble and rubble) size, which provides interstitial spaces, would likely reduce egg predation (Nester and Poe 1984). Feeding competition between larval lake whitefish and lake herring has been reported by Savino and Hudson (1995).

Neither lake whitefish nor the phylogenetically-related cisco (*C. artedii*) are able to spawn two years in succession because of the energetic cost involved in anadromous migrations of these species (James Bay watershed, Lambert and Dodson 1990*a*, 1990*b*). Lambert and Dodson (1990*b*) tested the hypothesis that the species-specific costs of migration differentially affected reproductive effort and somatic cost of reproduction in sympatric anadromous populations of cisco and lake whitefish of James Bay. The somatic energy increase in reproductive female lake whitefish was estimated to be 103% lower than the somatic energy increase of non-reproductive female, and much higher (121%) for cisco (Lambert and Dodson 1990*b*).

An energetically-costly behavior involving an extended (6-10 weeks) freshwater residence (without feeding) has been observed in anadromous lake whitefish of James Bay (Bernatchez and Dodson 1987). A majority of these anadromous fish migrate 27 km upstream, traversing about 6 km of rapids from mid August to mid September when the temperature drops to about 10 °C. When tested in the laboratory, the maximum oxygen consumption, aerobic scope of activity, and maximum sustained swimming speed for lake whitefish were maximal at 12 °C and minimal at 5 °C. Swimming endurance decreased logarithmically with swimming speed and was reduced at low temperature. These results suggest that migrating at low temperature may increase potential mortality before spawning.

Populations

Morrow (1980) has pointed out that "within each of the Great Lakes and probably in most large lakes, the lake whitefish form more or less separate populations." These populations are usually characterized by different growth rates, rather than by morphological differences. Sympatric populations of dwarf and normal lake whitefish commonly occur in north temperate and sub-Arctic lakes (Vuorinen et al. 1993; Chouinard et al. 1996; Trudel et al. 2001). The dwarf forms have a much lower growth rate, mature earlier and have a shorter life span than the normal forms. Interestingly, the dwarf form of whitefish is not found in lakes when cisco (*Coregonus artedi*) are present (Pigeon et al. 1997; Doyon et al. 1998), possibly due to competitive exclusion (Trudel et al. 2001). Studies conducted by Trudel et al. (2001) showed that the energy budgets of dwarf whitefish and cisco were similar, both consumed on average 40-50% more food than normal whitefish. But the conversion efficiency of both dwarf whitefish and cisco was 2-3 times lower than that of normal whitefish. Based on their results, these authors concluded that earlier maturation and shorter life span of dwarf whitefish may be due to their higher metabolic rates. By quantifying the extent of genetic (mitochondrial DNA and enzyme loci) and morphological differences between dwarf and normal whitefish subpopulations in Lac de L'Est, Quebec, Bernatchez et al. (1988) and Chouinard et al. (1996) showed significant differences in haplotype and allele frequency distributions, confirming that whitefish maturing at small (dwarf) and normal sizes are structured into two distinct gene pools.

Critical factors for survival

This review identified a number of factors that are critical to the survival of lake whitefish. Presence or absence of ice during incubation is critical because egg survival is reported to be 9-fold greater under ice. Success of over-wintering egg survival and amount of food available during the first 7 weeks of life are major determinants of year-class success (Freeberg et al. 1990). In Lake Michigan, Freeberg et al. (1990) also observed that abundance of suitable prey (for example, zooplankton ranging in size between 0.7 and 1.1 mm in Grand Traverse Bay, Lake Michigan) was critical for the survival of YOY. Altered temperature regimes can interfere with maturation and spawning migrations because both are stimulated by temperature (Edsall 1999*a*, 1999*b*).

Such changes in temperature can asynchronize the timing of hatching with the availability of appropriate-size food organisms (Ford et al. 1995). Reduced water levels can expose spawning substrates and make them unavailable to spawning fish (Gaboury and Patalas 1984). On the other hand, increased water level also has the potential to impact reproductive success by relocating depths preferred by lake whitefish.

Information deficiencies

Biological studies on lake whitefish in northern Canada have been largely conducted in the Mackenzie River drainage. As such, there is very little information concerning other northern populations of lake whitefish in Canada (Evans et al. 2002). Some of the major information deficiencies are with regard to migratory patterns in Arctic tundra watersheds, fidelity to spawning grounds and spawning substrates, trophic interactions, over-wintering refugia and habitat requirements under ice.

Table 8. Habitat and biological characteristics of lake whitefish, *Coregonus clupeaformis* (adapted from Ford et al. 1995).

Eggs	
Temperature	0-12 °C
Incubation Temperature	Optimum 4.0-6.0 °C
Incubation Time	42-182 days; 65 days at optimum temperature
Dissolved oxygen	Recommended >8.0 mg/L
pH	
Current Velocity	Recommended <scour velocity
Substrate	Sand, gravel, cobble, boulder
Cover	
Larvae and Juveniles	
Temperature	0-26.6 °C; optimum for growth 14 °C
Dissolved Oxygen	Recommended protection level >7.75 mg/L; lower lethal level <4.25 mg/L. The values are general critieria for freshwater fish (from Davis (1975)).
Habitat	Lakes
Preferred Depth	Varies with season
Preferred Current Velocity	
Substrate	Gravel, cobble, boulders
Cover	Vegetation and woody debris
Turbidity/TSS	<10 mg/L TSS
Primary Food	Benthic invertebrates
Secondary food	Zooplankton
Adults	
Temperature	Minimum preferred 8.0-14.0 °C; optimum for growth 7.75 °C
Dissolved Oxygen	Recommended >7.75 mg/L; short term minimum 4.25 mg/L. The values are general criteria for freshwater fish calculated by Davis (1975).
Habitat	Lakes
Preferred Depth	10 m-100+ m
Preferred Current Velocity	
Substrate	Only at the time of spawning
Cover	Darkness
Turbidity/TSS	<10 mg/L TSS
Primary Food	Benthic invertebrates
Secondary Food	Zooplankton
Spawning	
Spawning Habitat	Prefer shallow riffle areas in rivers or shallow beaches in lakes
Spawning Substrate	Sand, gravel, cobble, boulder
Nest Construction	None
Spawning Temperature	>8.0 °C
Spawning Depth	In rivers 0.1-1.0 m; large lakes 3.0-30.0 m; small lakes 1.0-5.0 m
Spawning Current Velocity	
Fecundity	8,800-35,495 eggs/female
Age at Maturity	Males 2-13 y; females 3-13 y

Table 9. Matrix indicating state of information on the biology and ecology of lake whitefish (*Coregonus clupeaformis*) in the Canadian Arctic.

Topics			**Life Cycle Stages**			
			Eggs	Larvae	Juveniles	Adults
Ecology and Habitat Requirements	Critical Habitat	Summer	n/a	**L**	**L**	**H**
		Winter	n/a	**L**	**L**	**L**
	Reproduction	Timing	n/a	n/a	n/a	**H**
		Habitat	n/a	n/a	n/a	**H**
	Rearing and Refugia	Summer	n/a	**L**	**L**	**M**
		Winter	n/a	**L**	**L**	**M**
	Food		n/a			**H**
	Water	Depth and Velocity				**M**
		Quality				
Behavior	Feeding		n/a			**H**
	Predation and Predator Avoidance		**M**			**M**
	Migration		n/a	**L**		**M**
Physiology/Biochemistry	Growth		n/a	**L**	**L**	**M**
	Function[4]					**L**
Trophic Linkages[1] and Bioenergetics[2]			**L**	**L**	**L**	**M**
Populations[3]			n/a			**M**
Human Impacts						

H = High; M = Medium; L = Low; n/a = not applicable. Blank boxes indicate no information available.
1 = Trophic linkages among or between species; 2 =Bioenergetics (balance between energy intake in the form of food and energy utilization by animals for life-sustaining processes) includes role of external factors such as global warming and pollution; 3 = Populations includes limiting factors and vulnerability to environmental changes; 4 = Ability to adapt/tolerate environmental changes.

Table 10. Matrix indicating state of information on the biology and ecology of lake whitefish (*Coregonus clupeaformis*) in areas south of 60°N.

Topics			**Life Cycle Stages**			
			Eggs	Larvae	Juveniles	Adults
Ecology and Habitat Requirements	Critical Habitat	Summer	n/a	L	L	L
		Winter	n/a			L
	Reproduction	Timing	n/a	n/a	n/a	H
		Habitat	n/a	n/a	n/a	H
	Rearing and Refugia	Summer	n/a			M
		Winter	n/a			L
	Food		n/a	L	L	H
	Water	Depth and Velocity	L	L	L	L
		Quality	L	L	L	L
Behavior	Feeding		n/a	L	L	H
	Predation and Predator Avoidance		L			M
	Migration		n/a		L	H
Physiology/Biochemistry	Growth		n/a			
	Function[4]			L	L	L
Trophic Linkages[1] and Bioenergetics[2]						
Populations[3]			n/a			M
Human Impacts						L

H = High; M = Medium; L = Low; n/a = not applicable. Blank boxes indicate no information available.
1 = Trophic linkages among or between species; 2 = Bioenergetics (balance between energy intake in the form of food and energy utilization by animals for life-sustaining processes) includes role of external factors such as global warming and pollution; 3 = Populations includes limiting factors and vulnerability to environmental changes; 4 = Ability to adapt/tolerate environmental changes.

ROUND WHITEFISH *(Prosopium cylindraceum)*

The habitat and biological characteristics of round whitefish are summarized in Table 11. Tables 12 and 13 provide a qualitative assessment of the state of knowledge about the biology of round whitefish in the Arctic and areas south of 60°N respectively.

Distribution

This species has widespread distribution in northern North America and into northeastern Asia. In North America, it is distributed from the New England states and maritime provinces of Canada to Alaska (McPhail and Lindsey 1970; Scott and Crossman 1973; Morrow 1980). It is found throughout the Northwest Territories and Nunavut, from Great Slave Lake throughout the Mackenzie River valley in Nunavut eastward to Keewatin district (McPhail and Lindsey 1970). It has been reported in brackish waters of the Mackenzie, Coppermine, and Churchill Rivers and off Clearwater in Quebec (McPhail and Lindsey 1970).

Round whitefish exhibits both lacustrine and adfluvial life history types and is reported to have riverine populations in some regions (Normandeau 1969; Bryan and Kato 1975; Morin et al. 1982).

Biology of life history stages

Adults: Adults are commonly found in shallows of lakes, ponds, slow flowing rivers and streams as well as in brackish waters (Normandeau 1969; McPhail and Lindsey 1970; Bryan and Kato 1975; Morin et al. 1982). These fish commonly occur in waters between 7 and 22 m deep with substrates characterized by rocks and boulders (Normandeau 1969; McPhail and Lindsey 1970). In Great Bear Lake, they prefer areas with currents and show preference for outlets of the lake (Kennedy 1949, cited in Richardson et al. 2001). Adults almost exclusively feed on small benthic invertebrates (Scott and Crossman 1973; Armstrong et al. 1977).

Larvae and juveniles: Young are found on the bottom of rocky areas and gravel in water depths of 1.5 to 5 m (Normandeau 1969; Goodyear et al. 1982, cited in Richardson et al. 2001). In laboratory experiments, regardless of light conditions, larvae preferred the bottom 10 cm of a test column and preferred sandy substrate (McKinley 1983, 1984).

Eggs: The eggs are broadcast over the substrate and incubate for 4-5 months under ice, hatching from March to May (Goodyear et al. 1982, cited in Richardson et al. 2001) most likely depending on the temperature. The eggs are orange in colour and unfertilized ova of the lacustrine New Hampshire population ranged between 2.4 and 2.9 mm, but after a few hours they swell to 3.3 to 4.6 mm (Normandeau 1969). In Alaska, unfertilized eggs are reported to be between 1.0 and 1.8 mm in diameter, but after absorption of water they swell to 3.0 to 5.0 mm (Furniss 1974).

Ecology of life history stages

Round whitefish have lower fecundity and larger egg size than other coregonines such as cisco (*C. artedii*) and lake whitefish (*C. clupeaformis*) (Morin et al. 1982).

Habitat use: According to McPhail and Lindsey (1970) and Scott and Crossman (1973), round whitefish usually inhabit shallow areas of lakes, ponds, and slow-flowing rivers and streams and can also be found in brackish waters. At the southern edge of their range, however, they are usually found in deep lakes. In Lake Michigan, they are commonly found at depths between 7 and 22 m, but could occasionally be found at depths up to 59 m (Becker 1983, cited in Bradbury et al. 1999).

Juveniles in the Susitna River (Alaska) migrate to the lower river for rearing during the first year and are reported to use high-turbidity waters as well as objects such as cobble, boulders, and overhanging riparian vegetation in calm waters for cover (Suchanek et al. 1984) preferentially or accidentally – a very significant fact for presence is not equated with optimal quality of water and habitat (Sundet and Wenger 1984, cited in Evans et al. 2002).

Spawning: This is reported to occur from autumn to early winter (Scott and Crossman 1973). In the north (Nueltin Lake, NWT) it can occur as early as October (Harper 1948, cited in Bryan and Kato 1975). In interior Alaska, spawning occurs in late September and October. Spawning appears to be an annual affair with many fish spawning in successive years (McCart et al. 1972, cited in Morrow 1980). Typically, spawning takes place at temperatures slightly above 0 °C (Morrow 1980; Bryan and Kato 1975). According to Scott and Crossman (1973), round whitefish spawn in lakes as well as rivers. Round whitefish have been reported to spawn in both fast and slow current areas (Normandeau 1969; Bryan and Kato 1975) but Bryan and Kato (1975) observed that round whitefish eggs were more commonly found in areas of high current velocity.

Upstream spawning migration has been observed near Nueltin Lake (at the border of Manitoba and NWT) in October (Harper 1948, cited in Scott and Crossman 1973). Normandeau (1969) has reported that round whitefish in New Hampshire (Newfound Lake, Bristol) arrive on the spawning grounds during the last two weeks of November. Here males arrive earlier than females and both sexes stop feeding during pre-spawning and spawning activities. No parental care was observed for eggs and young. We were unable to find any information with respect to the fidelity of round whitefish to spawning grounds. Although round whitefish have been observed to spawn over a variety of substrates ranging from silt and emergent vegetation to gravel and boulder, they spawn most commonly over gravel or rubble substrate (Normandeau 1969; Bryan and Kato 1975).

Spawning typically occurs in shallow waters <1 m deep (Normandeau 1969; Bryan and Kato 1975), but can also occur at depths between 5 and 10 m (Haymes and Kolenosky 1984, cited in Bradbury et al. 1999).

Eggs are broadcast over the substrate and incubate for 4-5 months under ice, hatching from March to May (Goodyear et al. 1982, cited in Richardson et al. 2001) most likely depending on the temperature. Females produce 1,000 to 12,000 eggs with the average between 5,000 and 6,000 (Morrow 1980). Egg size varies with locality. In southern locations (e.g. New Hampshire) of the distributional range average egg diameter may be about 0.27 cm (Normandeau 1969). In northern areas such as Alaska it is reported to be only 0.1 to 0.18 cm in diameter (Morrow 1980).

The incubation period has been reported to be around 140 days in New Hampshire and is believed to be similar in northern latitudes (Morrow 1980). Young hatch as sac fry and the yolk is absorbed in two to three weeks at which time they leave the spawning ground (Morrow 1980).

Rearing: After hatching (usually in April in temperate lakes), young are reported to remain on the bottom in areas with sand, rock, and gravel substrates at depths 1.5 to 4.5 m (Normandeau 1969). They are reported to position themselves at about 5-20 mm from the substrate (Evans et al. 2002). They disperse from the spawning area within 2-3 weeks (Morrow 1980, cited in Bradbury et al. 1999).

Food supply: Round whitefish appear to be opportunistic bottom feeders, almost exclusively ingesting small benthic invertebrates (Scott and Crossman 1973; Armstrong et al. 1977). In several northern lakes, a diet including caddisfly larvae and pupae, chironomid larvae, and small gastropods has been reported (McPhail and Lindsey 1970). In Lake Michigan, leeches, oligochaetes, smelt (*Osmerus mordax*) and fish eggs have also been reported in their diet, but in all seasons they most frequently feed on snails, and midge larvae (Armstrong et al. 1977).

Frequently, round whitefish have been observed to feed heavily on eggs of other fish species including lake trout, chum salmon, shad and suckers (McPhail and Lindsey 1970; Magnin et al. 1978).

Behavior

Migration: Upstream spawning migration of round whitefish in rivers has been reported to occur in late October (McPhail and Lindsey 1970). Although inshore and upstream migrations of round whitefish occur (Normandeau 1969), round whitefish do not display the characteristic “concentrated migratory” behavior of other coregonines such as cisco (Morrow 1980). They are reported to swim in pairs during spawning (a single male and a single female (Normandeau 1969)). Round whitefish do not build redds and eggs are broadcast in shallow waters which is consistent with the behavior of other whitefish species (Bryan and Kato 1975).

Physiology and biochemistry

Growth, fitness and survival: Growth rate varies considerably and reflects latitude and temperature differences (Scott and Crossman 1973; Morrow 1980; Magnin and Clement 1982). In Lake Michigan, round whitefish are reported to grow rapidly, reaching a length of 50 cm in 7 years. In northern latitudes (e.g. Brooks Range, Alaska), however, this length is not achieved until age 12. Sexual maturity in Alaska is reached at age 7 (Morrow 1980), but in more southerly locations round whitefish mature much earlier at about age 5 (Mraz 1964; Mackay and Power 1968; Jessop and Power 1973; Armstrong et al. 1977).

Studying the life history variations of anadromous coregonines in La Grande River, James Bay, Morin et al. (1982) found that relative to cisco (*C. artedii*), round whitefish displayed reproductive patterns more typical of harsh northern environments (i.e. increased ages at maturity, fewer but larger young, longer life expectancy, and smaller reproductive efforts).

Age at maturity: Stein et al. (1973) reported that age at maturity was 8 years. An earlier report (Kennedy 1949, cited in Richardson et al. 2001) documented that round whitefish mature between ages 6 and 7. In a New Hampshire lake (Newfound Lake) it has been reported that they attain sexual maturity in their 4th or 5th year of life (Normandeau 1969).

Trophic linkages

Little research has been done with respect to trophic linkages and trophic status of various round whitefish populations other than casual observations of predators and prey. Sandercock (1964) provided evidence for apparent interspecific competition between round whitefish and lake whitefish in Algonquin Park (Ontario). He reported depressed growth rates for round whitefish when lake whitefish were present in lakes and attributed this depressed growth rate to greater efficiency of lake whitefish in capturing planktonic organisms. Round whitefish have been considered by several authors to be a significant predator on the eggs of lake trout (Morrow 1980).

Normandeau (1969) has reported lake trout predation on round whitefish, but the species was not a major item in the lake trout diet. This author also reported that brown bullheads and burbot also consumed round whitefish eggs in Newfound Lake, New Hampshire. According to Scott and Crossman (1973), it is possible that other fish (such as yellow perch) may also prey on round whitefish eggs.

Critical factors for survival

Habitats critical to survival (such as over-wintering refugia and spawning areas) are similar for all fish species inhabiting the Arctic. In addition, interspecific competition is also critical since round whitefish share habitat with other predatory fish species such as lake trout.

Information deficiencies

Like most other Arctic fish, there is a significant lack of basic information on habitat requirements of round whitefish in the Arctic. Information is particularly lacking for life history traits, over-wintering refugia, migrations and movements, site fidelity for spawning grounds or to other areas such as summer feeding grounds, habitat requirements for various life stages, and trophic interactions and status of lacustrine, adfluvial, and fluvial populations. Spawning migrations have been reported in rivers (Morrow 1980), but we found no studies on their fidelity to spawning sites.

Table 11. Habitat and biological characteristics of round whitefish, *Prosopium cylindraceum*.

Eggs	
Temperature	
Incubation Temperature	
Incubation Time	4-5 months under ice
Dissolved oxygen	
pH	
Current Velocity	Eggs reported from areas of both slow and fast current
Substrate	Most abundant on gravel, but also found on silt and boulders
Cover	
Larvae and Juveniles	
Temperature	
Dissolved Oxygen	
Habitat	Bottom in rocky and gravelly areas
Preferred Depth	1.5 to 5 m deep, in lab. experiments preferred 10 cm
Preferred Current Velocity	
Substrate	Gravel, rubble
Cover	
Turbidity/TSS	
Primary Food	
Secondary Food	
Adults	
Temperature	
Dissolved Oxygen	
Habitat	Shallow lakes, ponds, rivers, streams and brackish waters
Preferred Depth	7 to 22 m deep
Preferred Current Velocity	Specific values not available. In rivers prefer areas with current but also spawn in shallows of lakes (e.g. Great Bear Lake (Kennedy 1949)
Substrate	Rocks, boulders
Cover	
Turbidity/TSS	
Primary Food	Opportunistic bottom feeders; insect larvae, gastropods. Leeches, oligochaetes
Secondary Food	Fish eggs
Spawning	
Spawning Habitat	Lakes, rivers, streams
Spawning Substrate	Most commonly over gravel or rubble
Nest Construction	None, broadcast spawners
Spawning Temperature	Slightly above 0 °C
Spawning Depth	Commonly <1 m deep, but may be 5-10 m deep
Spawning Current Velocity	
Fecundity	5,000 to 6,000 eggs/female
Age at Maturity	Commonly at 4 to 5 years of age, but in the north up to 8 years

Table 12. Matrix indicating state of information on the biology and ecology of round whitefish (*Prosopium cylindraceum*) in the Canadian Arctic.

Topics			**Life Cycle Stages**			
			Eggs	Larvae	Juveniles	Adults
Ecology and Habitat Requirements	Critical Habitat	Summer	n/a	**L**	**L**	**L**
		Winter	n/a			
	Reproduction	Timing	n/a	n/a	n/a	**M**
		Habitat	n/a	n/a	n/a	**M**
	Rearing and Refugia	Summer	n/a	**L**	**L**	**M**
		Winter	n/a			
	Food		n/a			**L**
	Water	Depth and Velocity				**L**
		Quality	**L**	**L**	**L**	**L**
Behavior	Feeding		n/a			**L**
	Predation and Predator Avoidance					**L**
	Migration		n/a			**L**
Physiology/Biochemistry	Growth		n/a			**L**
	Function[4]					
Trophic Linkages[1] and Bioenergetics[2]						**L**
Populations[3]			n/a			
Human Impacts						

H = High; M = Medium; L = Low; n/a = not applicable. Blank boxes indicate no information available.
1 = Trophic linkages among or between species; 2 = Bioenergetics (balance between energy intake in the form of food and energy utilization by animals for life-sustaining processes) includes role of external factors such as global warming and pollution; 3 = Populations includes limiting factors and vulnerability to environmental changes; 4 = Ability to adapt/tolerate environmental changes.

Table 13. Matrix indicating state of information on the biology and ecology of round whitefish (*Prosopium cylindraceum*) in areas south of 60°N.

Topics			**Life Cycle Stages**			
			Eggs	Larvae	Juveniles	Adults
Ecology and Habitat Requirements	Critical Habitat	Summer	n/a	L	L	L
		Winter	n/a			
	Reproduction	Timing	n/a	n/a	n/a	M
		Habitat	n/a	n/a	n/a	M
	Rearing and Refugia	Summer	n/a	L	L	M
		Winter	n/a			
	Food		n/a			L
	Water	Depth and Velocity				L
		Quality				L
Behavior	Feeding		n/a			
	Predation and Predator Avoidance					L
	Migration		n/a			L
Physiology/Biochemistry	Growth		n/a			M
	Function[4]					
Trophic Linkages[1] and Bioenergetics[2]						L
Populations[3]			n/a			L
Human Impacts						

H = High; M = Medium; L = Low; n/a = not applicable. Blank boxes indicate no information available.
1 = Trophic linkages among or between species; 2 = Bioenergetics (balance between energy intake in the form of food and energy utilization by animals for life-sustaining processes) includes role of external factors such as global warming and pollution; 3 = Populations includes limiting factors and vulnerability to environmental changes; 4 = Ability to adapt/tolerate environmental changes.

BURBOT (*Lota lota*)

The habitat and biological characteristics of burbot are summarized in Table 14. Tables 15 and 16 provide a qualitative assessment of the state of knowledge about the biology of burbot in the Arctic and areas south of 60°N respectively.

Distribution

Burbot is the only freshwater species of the otherwise marine family of cods (*Gadidae*). It has a circumpolar distribution from the British Isles eastward across Europe and Asia to the Bering Strait (Berg 1949, cited in McPhail 1997), and on the North American side from Bering Strait eastward from Alaska to New Brunswick (McPhail and Lindsey 1970; McPhail 1997). Survival in refugia during glaciation suggests that there are genetically-distinct forms (McPhail 1997). Snyder (1998) has provided evidence of two distinct genetic forms of burbot based on larval morphology, and Paragamian and MacKinlay (1998) and Paragamian et al. (1999) have provided evidence of genetic groups of burbot based on differences in mitochondrial DNA.

Burbot occur throughout the Northwest Territories and Nunavut exclusive of the northernmost areas, and are reported to be absent from Arctic islands (McPhail and Lindsey 1970; Scott and Crossman 1973; Tripp et al. 1981).

The species is usually found in deepwater lakes but is also found in rivers (McPhail 1997). As such, burbot are represented by riverine, lacustrine and adfluvial types. Occasionally, individuals are found in brackish waters. Feeding and spawning migrations into rivers of lacustrine populations have been reported. They are primarily adapted to a benthic and nocturnal habit (Scott and Crossman 1973; Bond and Erickson 1991).

Biology of life history stages

Adults: Adults are found in deep lakes or cool rivers and reservoirs associated with mountainous areas (McPhail 1997). In lakes, adult burbot are strongly benthic and they appear to occupy habitats in relation to thermal requirements (McPhail 1997). Preferred temperatures are reported to be between 10 °C and 12 °C and burbot appear to avoid temperatures above 13 °C (Hackney 1973, cited in McPhail 1997). It is, therefore, not surprising that in summer burbot are usually found below the thermocline in lakes. In Lake Superior burbot regularly occur in waters as deep as 300 m where they are reported to construct extensive burrows in the substrate. In some lakes, burbot move into shallow areas or even into rivers in late fall where they feed prior to spawning.

The use of habitats in rivers by burbot is unclear according to McPhail (1997). In northern rivers where temperature rarely exceeds 18 °C, burbot are common (Bishop 1975; Breeser et al. 1988), but they are relatively uncommon in southern rivers where summer temperature exceeds 20 °C. In northern rivers, burbot are generally associated with main channels, and the "inhospitable environment of the glacial rivers during summer, when flows, turbidity, and scouring are at high levels, apparently had little effect on burbot" (observations on 10 tagged fish by Breeser et al. 1988). Some authors

(Chen 1969; Hatfield et al. 1972, both cited in McPhail 1997) have observed burbot in turbid waters. Burbot often enter tributaries of northern rivers in the fall, and in the Mackenzie delta burbot are known to enter brackish waters but return to rivers in the fall (McPhail 1997).

Over 80% of the adult burbot diet is reported to consist of fish (refer to McPhail 1997). In North America, burbot feed on a variety of fish species including lamprey, whitefish, grayling, pike, sucker, minnows, stickleback, trout-perch, yellow perch, sculpin and other burbot (McPhail 1997). Seasonal shifts in burbot diet (from fish and crustaceans in winter to *Mysis* and *Pontoporeia* in summer) has been reported (Bailey 1972).

In northern rivers, adult burbot are normally associated with main channels, and they have been found in waters with "moderate" to "high" turbidity, low current velocities (<46 cm/s), and shallow depths (<76 cm) (Suchanek et al. 1984). According to Ford et al. (1995) "little is known about the turbidity levels that are harmful to burbot" but these authors provide anecdotal information suggesting that "burbot have been observed in some Alaskan rivers when turbidity has reached up to 1,000 NTU (nephelometric turbidity units). Chen (1969, cited in Ford et al. 1995) stated that in the Tanana and Yukon Rivers of Alaska, burbot "are more abundant in silty main rivers than in the smaller, clear tributaries." However, quantitative information that indicated the species tolerance or resistance to suspended sediment, or their preferences for turbid conditions was not obtained through the search of literature.

Larvae and juveniles: Like most gadoid species burbot larvae are also pelagic and positively phototactic. While the juveniles are active at night and have a benthic lifestyle in the littoral zone (Ryder and Pesendorfer 1992), burbot larvae in lakes are pelagic. Upon hatching in riverine situations, the larvae drift passively in the water column to settle in quieter waters downstream which serve as nursery and rearing areas (McPhail 1997). Their swimming performance improves as they grow (Clady 1976; Ghan and Sprules 1991; Ryder and Pesendorfer 1992). The maximum concentration of burbot larvae in Lake Michigan was observed in March to mid June at depths between 3.0 and 7.5 m (Mansfield et al. 1983). Burbot larvae in relatively southern lakes (e.g. Lake Oneida, New York) are reported to abandon their pelagic mode of life in June (Clady 1976; Ghan and Sprules 1991), suggesting a habitat shift from pelagic to demersal in early summer (McPhail 1997). The shift appears to occur at sizes above 15 mm in Lake Oneida (Ghan and Sprules 1993). A more detailed description of habitat shifts from littoral to profundal zones of lakes is provided below regarding habitat use.

Eggs: Burbot eggs are not adhesive and have been variously described as demersal, semibuoyant and buoyant (McPhail 1997). Egg size appears to vary considerably. For example, McPhail (1997) reported that the egg diameter in eastern North American populations of burbot is generally greater (for example 1.7 mm in Lake Erie) than those of western North America and Alaska (for example 0.53 mm in Alaska).

Ecology of life history changes

Habitat use: Although some populations in deep, cold, northern rivers have been reported, burbot is essentially a deep-water, lake-dwelling fish (Scott and Crossman 1973; McPhail 1997). In large lakes, it utilizes several habitats during ontogenesis:

larvae are typically pelagic (Ghan and Sprules 1993), juveniles inhabit littoral zones (Carl 1995), and adults mostly use the cold profundal zone (Scott and Crossman 1973; McPhail 1997). The underlying mechanisms of these ontogenetic habitat shifts are not well understood, although avoidance of conspecific predator risk has been speculated upon (Fischer 1999; Hofmann and Fischer 2001).

Little is known about movements of and habitat use by riverine populations of burbot (Slavik and Bartos 2002). They are believed to be sedentary except during spawning periods in late fall when they move up small tributaries to spawn (Morrow 1980; Breeser et al. 1988). According to Breeser et al. (1988), burbot occupying rivers in the Yukon exhibited extensive movements within the river throughout the year. With the approach of winter when the river discharge is low and water temperature begins to fall, burbot migrate to deeper waters for over wintering (Breeser et al. 1988; Slavik and Bartos 2002). There is a paucity of information on the larval ecology of fluvial populations of burbot.

Young of the year burbot occupy habitats under stones and debris in shallow bays during daylight and leave these areas to forage at night (Ryder and Pesendorfer 1992). YOY of the river populations have been observed to occupy weed beds, areas under rocks and debris during the daylight (Hanson and Qadri 1980).

Adult burbot in lakes are strongly benthic in habit, and habitat selection appears to be dependent on temperature. The preferred feeding temperature for burbot is reported to be between 10 °C and 12 °C (littoral zone), but they avoid water temperature above 12 °C (Hackney 1973, cited in McPhail 1997; Hofmann and Fischer 2002). The emigration of 12-16 cm long burbot from the littoral zone in the summer is probably stimulated by a rise in temperature (Fischer and Eckmann 1997, cited in Hofmann and Fischer 2002). Fischer (2000) measured respiratory responses of burbot to different types of substrates and showed that lack of adequate shelter may substantially affect metabolism and somatic growth rates.

Spawning: Burbot spawn in lakes and rivers during winter months (January to March) under ice, or in areas where rivers remain open (McPhail and Lindsey 1970; Scott and Crossman 1973; McPhail 1997). In general, however, burbot spawn in winter or early spring (December to early March) often under ice. But, in Kooteney Lake (BC) they are reported to begin spawning in early April and continue into late May or June (Martin 1977, cited in McPhail 1997). Fluvial burbot make migrations as long as 125 km to reach their spawning grounds (Breeser et al. 1988). In late fall (November), major spawning migrations have been reported in the Slave River (McLeod et al. 1985, cited in Evans et al. 2002).

Spawning in lakes occurs in nearshore, shallow areas 1.5 to 10 m deep, or over shallow, offshore reefs and shoals. Deepwater spawning in the Great Lakes has also been reported (Clemens 1951, cited in McPhail 1997). Spawning usually occurs above substrates of coarse gravel or cobbles which are relatively free of silt.

In rivers, burbot spawn in low velocity areas in main channels (Breeser et al. 1988), and in side channels behind deposition bars (Sorokin 1971). The substrate where spawning has occurred in rivers, has been of fine gravel, sand, or even fine silt (McPhail 1997).

The temperature at the time of spawning has been reported to lie between 1 °C and 4 °C (refer to McPhail 1997). A shift in temperature from 0 to 2.5 °C is reported to delay spawning by 14 days.

Spawning usually involves more than just a pair of fish. Several males (up to 12) surround one or two females in a "writhing ball" (Cahn 1936, cited in McPhail 1997). The eggs are broadcast into the water column well above the substrate. Eggs initially drift with currents and waves but eventually sink to the bottom, settling into interstices in the substrate (Sorokin 1971).

Batch fecundity is very high. Egg estimates range from 6,300 (Miller 1970, cited in McPhail 1997) to 3,477,699 (Roach and Evenson 1993, cited in McPhail 1997), which is not uncommon in cods (Bailey 1972). Fecundity in burbot is, however, quite variable and can vary significantly between lakes in the same geographic area (Boag 1989, cited in McPhail 1997). Fecundity is positively correlated with size and age, but the effect of age on fecundity is not as pronounced in burbot as in many other fish (McPhail 1997).

Not all adult fish spawn each year. In the Tanana River (Alaska) about 15% of the females and 17% of the males aged 7 or older would not have spawned in the year they were captured (Evenson 1990, cited in McPhail 1997). Similarly, in Alberta, 5% of the adults in Cold Lake and 1.5% of adults in Lac Sainte Anne did not spawn in consecutive years (Boag 1989, cited in McPhail 1997). Pulliainen et al. (1992) found non-spawning burbot on the northern coast of the Bothnian Bay, Finland (a shallow sea area with salinity below 2‰) and speculated that its causes could include "infection or a resting year phenomenon."

Egg size varies considerably (0.53 mm in Alaska, 1.7 mm in Lake Erie) and it appears that egg size in eastern North American populations is larger than in western North American populations (McPhail 1997).

Information on conditions for successful incubation of eggs is sparse. The optimum development temperature is reported to be between 4 °C and 7 °C. But, fertilized egg mortality increases at water temperatures below 4 °C. Jager et al. (1981), however, reported that the optimum temperature regime for development ranges between 1 °C and 7 °C and emphasized that on either side of 4 °C, zygote mortality sharply increases. McPhail (1997), pointed out that most of the natural incubation regimes inferred for North American and European populations are below 4 °C.

Burbot eggs have been reported to hatch under ice cover (Ryder and Pesendorfer 1992), and by mid April the larvae have been observed feeding on planktonic organisms such as rotifers, copepods, and cladocerans in eastern North America (Ryder and Pesendorfer 1992; Ghan and Sprules 1993).

Larval size depends on egg size, i.e. larger eggs produce larger larvae (McPhail 1997). Length at hatching, therefore, varies and has generally been reported to be around 3 mm (Mansfield et al. 1983).

Larval densities shortly after hatching can be quite high (up to $15/m^2$ of surface area) but within four weeks the larval density decreases to about $1/m^2$ (Ghan and Sprules 1991), suggesting high mortality and/or dispersal. According to Mansfield et al. (1983), the newly-hatched burbot larvae in Lake Michigan are more buoyant than eggs and have little fin development. As such, they are susceptible to movement by water currents and can be found throughout the water column. Mansfield et al. (1983) reported that in Lake Michigan, burbot larvae were often collected in water wherein the temperature ranged between 6 °C and 12 °C.

Rearing: Upon hatching, sac fry are found mainly in the pelagic zone and are reported to congregate over sand and rubble substrates (McPhail 1997).

Little is known about the larval ecology of burbot's fluvial populations. McPhail (1997) provided anecdotal information to suggest that quiet waters are often found downstream of spawning areas and may serve as nursery grounds for this stage in the life cycle.

Field observations in Oneida Lake, New York suggest that exogenous feeding begins about 5 days after hatching, and the first food items are small planktonic organisms such as rotifers (Ghan and Sprules 1993). Ryder and Pesendorfer (1992) reported copepods and cladocerans as the first food items of fry in a Precambrian Shield lake.

Two ontogenetic feeding stages have been suggested by Hartmann (1983, cited in McPhail 1997): at lengths between 5 and 14 mm, the number of particles in the gut increases with larval size, but in sizes from 15 mm to transformation (i.e. about 30 mm), the number of particles in the gut remains relatively constant while particle size increases with fish size. Larval growth has been reported to be rapid in May and June but decreases in August (Ryder and Pesendorfer 1992).

At the fingerling stage (lengths between 20 and 40 mm), YOY adopt a benthic mode of life (Ryder and Pesendorfer 1992; McPhail 1997). According to McPhail, this change in habit also marks a shift from crepuscular (twilight-related) to nocturnal activity. Upon becoming nocturnal, young burbot inhabit shallow areas under boulders, cobble or submerged vegetation (Ryder and Pesendorfer 1992; Ford et al. 1995).

Both juvenile and adult burbot move offshore to deeper waters of the lake hypolimnion in early summer (Scott and Crossman 1973; Ford et al. 1995).

Food supply: The main food of young burbot consists of small benthic organisms (e.g. insect larvae, rotifers, etc. (McPhail 1997)). In the Yukon and Tanana (Alaska) Rivers, juvenile burbot feed mainly on insect larvae, especially plecopterans, ephemeropterans, and dipterans, and on slimy sculpin (Chen 1969; Thornburgh 1986, both cited in Ford et al. 1995). These authors also provided evidence that by the third or fourth year, burbot juveniles consume mainly fish but may also feed on crayfish as well as aquatic insects.

In southwestern Lake Superior, burbot of all sizes fed on fish and crustaceans (Bailey 1972). In this study, fish heavily dominated the diet of large burbot captured during the winter months (over 99% of the volume), but crustaceans (*Mysis* and *Pontoporeia* sp.) became increasingly important during summer months and by fall they contributed about 73% of the volume of the diet. The fish consumed by burbot in various lakes include cottids, coregonids, alewives, yellow perch, lake herring, and emerald shiners (refer to

Bailey 1972). In late fall, burbot consumed large quantities of coregonid eggs (59.7% by occurrence; 9.2% by volume). On the basis of the large quantities and variety of fish consumed by burbot, it would appear that this species is a major competitor of most of the fish in its habitat.

In laboratory experiments, newly-hatched larvae first ate phytoplankton and did not switch to copepod nauplii until the third day of exogenous feeding (Vatcha 1990, cited in McPhail 1997). It has been suggested that burbot larvae select the largest prey items they can engulf (Ghan and Sprules 1993).

Two ontogenetic feeding stages have been suggested by Hartmann (1983, cited in McPhail 1997): at lengths between 5 and 14 mm the number of particles in the gut increased with larval size, but from 15 mm to transformation at about 30 mm the number of particles in the gut remained relatively constant while particle size increased with fish size.

In a recent study of the feeding ecology of three piscivorous benthic fish species (burbot; perch, *Perca fluviatalis*; and pike, *Esox lucius*) in the sub-Arctic Pasvik watercourse (69°N) in Norway and Russia, Amundsen et al. (2003) found that ontogenetic diet shifts and resource partitioning patterns of these three piscivorous fish represented substantial food web complexity. Burbot showed a distinct ontogenetic niche shift, changing from a dominance in diet of zoobenthos to fish. Fish prey size was positively correlated to predator size in burbot and the other two species. A cluster analysis by these authors separated the different size groups of the three species into five functional feeding groups, most of them containing two or all three species. Similarly, Kahilainen and Lehtonen (2003) found that in Lake Muddusjaervi (a whitefish-dominated, sub-Arctic lake in northern Finland), burbot along with pike consumed more prey fish and took a wider range of prey sizes than other piscivores, and the prey length for all predators increased in relationship to predator length.

In rivers (e.g. Ottawa River), young burbot are found in weed beds, and under rocks and debris during the day where they feed on amphipods, mayflies, stoneflies and the young of other fish (Bishop 1975; Hanson and Qadri 1980, cited in McPhail 1997).

Behavior

Feeding and predator avoidance: In Lake Superior, burbot have been reported to construct sedimentary structures at depths ranging from 140 to 366 m (Boyer et al. 1989, 1990). The actual ecological significance of this trench building behavior is not known. However, it has been speculated that these trenches could be a vestigial predator-avoidance mechanism inherited from marine members of the cod family, or they are constructed to facilitate the capture of sculpin prey (Boyer et al. 1989).

Migration: It appears that except for spawning migrations, burbot in both rivers and lakes are relatively sedentary (McPhail 1997). In the Slave River, for example, some radio-tagged burbot moved upstream (40-280 km) but others remained in the area of release (Tallman et al. 1996*a*, 1996*b*). Spawning migrations occur in February in the Slave River, which according to Scott and Crossman (1973) is similar for most burbot populations.

Several lacustrine populations of burbot have been reported to be adfluvial and some evidence has been provided for spawning site fidelity (McPhail 1997). For example, from Columbia Lake, BC, burbot have migrated into Dutch Creek in February and to the same site over many years. However, when that site was channelized, the population shifted to another spring-fed tributary where the substrate was of fine gravel (McPhail 1997). Burbot tagged in Great Slave Lake did not show distinct patterns of movement (Keleher 1963), and two-thirds of the tagged fish were recovered within 10 km of the tagging site, although average time between release and capture was 581 days.

Seasonal movement of burbot along short channels and between lakes has also been observed (Kennedy 1940, cited in McPhail 1997). In Alaska, however, only limited movement of tagged fish between similarly-connected lakes was reported (Lafferty et al. 1990, cited in McPhail 1997). In contrast, fluvial populations of burbot have been recorded to migrate long distances to specific spawning sites, the longest being 255 km (Tripp et al. 1981, cited in McPhail 1997). Radio tagging of burbot in Tanana River, Alaska, revealed that small burbot showed no seasonal pattern to their movements, but the larger burbot made extensive migrations. The greatest such migrations occurred during periods coinciding with river freeze up and river ice out (Evenson 1993*a*, 1993*b*). This suggested a spawning migration and the author inferred the location of a number of spawning sites in the river and its tributaries. Breeser et al. (1988) observed the "longest" movement of burbot in Tanana River (Alaska) during the November-March period and attributed it to movement to spawning grounds. The movement at the ice-out period as observed by Evenson (1993*a*) could well be movement to summer feeding areas.

Other behavioral patterns: In northern lakes (such as Fielding and Paxon Lakes in Alaska), burbot undergo two major annual phase shifts in their activity. They become diurnal in October and nocturnal in February, and at the times of shift, the duration (about 11 h/day) of locomotor activity is the greatest (Müller 1969, 1973; Kroneld 1976, cited in Bernard et al. 1993). It has been speculated that changing photoperiod and changing water temperature stimulate burbot to move (Bernard et al. 1993).

In Lake Constance (47°40'0"N, Germany), burbot larvae (7-10 mm) "exhibit a distinct diel vertical migration behavior, which is typical for many gadoid species prior to settlement" (Miler and Fischer 2004). From the end of May onwards, these authors observed a significant diel vertical migration pattern of larval burbot whose abundance in the pelagic zone had peaked during April. While burbot larvae were distributed above as well as below the thermocline in the day time with the greatest abundance at water depths of between 30 and 50 m (below the thermocline), they were concentrated from 2 to 15 m (above the thermocline) during the night. The authors further observed that burbot larvae and their preferred food organisms performed temporally-synchronized diel vertical migrations in early summer. These diel vertical movements of burbot larvae and zooplankton have been considered as a possible behavioral adaptation against predation pressure by planktivorous fishes (such as lake whitefish in northern lakes (Miler and Fischer 2004)). It is also plausible that these vertical movements to capture prey in warmer waters followed by a retreat to colder waters facilitates the growth of individuals through the greater partitioning of energy towards growth rather than metabolic needs that would be higher in the warmer waters. Such diurnal, vertical, and coincident temperature shifts of rearing sockeye salmon juveniles (*Oncorhynchus nerka*) in lakes are well documented (e.g. Brett 1971; Clark and Levy 1988).

Physiology and biochemistry

Growth, fitness and survival: The life span of burbot varies considerably geographically. Generally, the northern populations contain older fish than do southern populations. For example, in Quebec individuals older than 7 years are rare in populations at 45°N, but in populations at 55°N most adults are 8-12 years old (Magnin and Fredette 1977, cited in McPhail 1997). In more northern areas 20-22 year old individuals have been reported (McPhail 1997). In lower Slave River, burbot matured later (age 5) and grew more slowly than other populations (Tallman et al. 1996*b*), which mature at age 3-4 (Scott and Crossman 1973).

Spawning of burbot "represents the greatest annual metabolic demand for the species" (Mustonen et al. 2002; Pääkkönen et al. 2003). These authors observed a significant decrease in body mass, relative weight of the livers, and glycogen concentration of the livers towards the end of spawning.

Burbot populations also demonstrate a great deal of variation in individual growth rates. For example, they reach an average length of 147 mm in their first year in Heming Lake (Manitoba) and 210 mm in Lake Erie (Ford et al. 1995). Although Kirillov (1988) reported that female burbot grew faster than males at least until age 4, others (Holcik and Nagy 1987, cited in Ford et al. 1995) found no difference in either growth rate or length-weight relationship between males and females in River Turiec (former Czechoslovakia).

Age at maturity: The age at sexual maturity varies both geographically and with sex. Burbot usually mature between 3 and 4 years of age, with males maturing a year or two earlier than the females (McPhail 1997). However, in northern latitudes (Northwest Territories, for example) they have been reported to mature at 5 years of age (Scott and Crossman 1973; Tallman 1996*a*, 1996*b*).

Tolerance and resistance: Although adult burbot are usually found in relatively low velocity areas along lake bottoms (Ford et al. 1995), they are also reported to tolerate high summer peak flows in glacial Tanana River in Alaska (Breeser et al. 1988). However, Dryden and Stein (1975, cited in Ford et al. 1995) reported that the maximum velocity barrier for burbot was 20 cm/s. Clearly, with such variations reported, there is a need for further research to determine the optimum current velocity for burbot (and for other species in areas where habitat manipulation might involve changes in current velocities).

Although adult burbot are reported to occur in turbid waters (Chen 1969; Hatfield et al. 1972, both cited in McPhail 1997), little is known about the turbidity levels which could be harmful to them. Burbot have been observed in waters with turbidity levels as high as 1,000 NTU (nephelometric turbidity units) in glacial rivers at the height of summer. It has also been reported that they are more abundant in the silty waters of the Yukon and Tanana Rivers than in smaller and clearer tributaries (Ford et al. 1995). This aspect requires further investigation to establish the optimum turbidity requirement in burbot habitat and the behavioral issues surrounding the use of such habitats which may include the use of turbid waters as cover to assist in food capture by this ambush predator.

According to Scott and Crossman (1973), the optimum temperature range for both juvenile and adult burbot lies between 15.6 °C and 18.3 °C and the upper tolerance limit is 23.3 °C. As such, in summer burbot occupy cooler hypolimnetic waters of deeper lakes and waters with a temperature between 10 °C and 12.5 °C (Kirillov 1988). Similarly, in a more recent study conducted in Poland on 2-month old burbot reared at 12, 15, 18 and 21 °C, the best growth (mass and length), and the highest survival (97%) was at 15 °C (Wolnicki et al. 2001, 2002).

Temperature is considered to be a factor that guides the ontogenetic habitat shift from shallow littoral areas to cold profundal waters in lakes (Hofmann and Fischer 2002). In laboratory thermal gradient experiments, Hofmann and Fischer (2002) studied the effect of acclimation temperature (AT) on preferred temperature (PT) and the critical thermal maximum and minimum of burbot. These authors found a nonlinear relationship between PT and AT in small (10-20 cm) burbot with the final temperature preferendum at 11.4 °C for small (10-20 cm) burbot. In larger (20-30 cm) burbot no such relationship was found and the final temperature preferendum was 14.2 °C. However, in both sizes of burbot a positive linear relationship was found between the critical thermal maximum temperature and the acclimation temperature. A critical thermal minimum of 3.3 °C was found for small burbot acclimated to 19.6 °C. Based on these laboratory results, the authors suggested that changes in water temperature in spring would act as a "trigger" for juveniles to leave the littoral zone and move to waters at preferred temperatures, assuming that suitable food resources were available within that location.

Impacts of human interventions: Overfishing, hydroelectric dams, gas pipeline construction, forest industry wastes, paper mills and climate change have been reported to negatively impact burbot populations. Overfishing has been reported to cause declines in the abundance of older burbot in Heming Lake, Manitoba (Lawler 1963, cited in Ford et al. 1995). Erosion of shoreline following flooding can lead to poor egg survival because of insufficient dissolved oxygen around eggs caused by sedimentation. On the other hand, drawdown of lakes reduced densities of invertebrates, which influenced the rearing habitats of burbot. In Bull Lake, Montana for example, the removal of water for irrigation caused a 50% decline in spawning and rearing habitat (Ford et al. 1995).

Trophic linkages

There is a broad diet overlap between burbot and other piscivorous species such as walleye, lake trout and bull trout (Bailey 1972; McPhail 1997), which suggests competitive interactions among these species. Comparing the numbers and growth rates of burbot in an unexploited population in a lake with an exploited lake trout population (Day 1983, cited in McPhail 1997), it was found that burbot numbers increased but growth rate decreased as lake trout numbers declined. However, in contrast, Carl (1992) observed no changes in burbot growth or numbers at a time when lake trout numbers increased by 20% and the author suggests that predation by planktivores such as lake herring on larval burbot may be responsible for this apparent larval "bottleneck" in burbot. Tolonen et al. (1999) found considerable diet overlap between burbot and whitefish (*Coregonus lavaretus*), particularly during the period of ice cover.

Cannibalism also appears to be an important control mechanism in burbot. Chen (1969, cited in McPhail 1997) observed that at some times of the year young burbot are a major food item of adults in the Tanana River (Alaska).

Populations

The presence of lacustrine, adfluvial and fluvial life-history forms within the same system (such as the upper Columbia River system) suggest that burbot may be divisible into genetically-discrete demes (a local, usually stable, population of interbreeding organisms of the same kind or species) or stocks (McPhail 1997). No substantive work has been done in this regard in North America but it needs to be investigated to determine if the life histories of burbot differ among ecological regions (McPhail 1997).

The density of burbot in lakes has been shown to be variable. In Alaskan lakes (where extensive stock assessment work has been conducted), adult (>450 mm) density estimates range from 0.24-21.9/ha surface area (refer to McPhail 1997). The highest adult densities (average 139, range 0-571/ha of lake bottom) were estimated in southwestern Lake Michigan (Edsall et al. 1993). Thus it would appear that burbot densities in Arctic lakes are lower than in southern lakes such as Lake Michigan.

Because of high batch fecundity of burbot, larval densities after hatching are high. In Lake Oneida (New York), densities shortly after hatching were up to $15/m^2$ of surface area, but within four weeks the larval density decreased to about $1/m^2$ (Ghan and Sprules 1991) suggesting high mortality. Ryder and Pesendorfer (1992) reported a similar rapid decline in the relative abundance of larvae from hatch to the shift to a benthic habit. Carl (1992), assuming a 1% survival from larvae to benthic settlement, estimated about 24 YOY per linear metre of suitable shoreline decline in larval density. It has been suggested that predation by planktivores on larval burbot may be responsible for the apparent low recruitment (Carl 1992). McPhail (1997) speculated that cannibalism may be an important population-control mechanism in burbot, for it has been noted by Chen (1969, cited in McPhail 1997) that "at some times of the year, young burbot are a major food items for adults in the Tanana River, Alaska."

Critical factors for survival

Like many other fish, burbot pass through certain "bottlenecks" during their life cycle which can be critical for survival (Carl 1998), such as the shift from the pelagic larval phase to benthic life as reported by Miler and Fischer (2004). Any factors (such as availability of food and substrates) limiting this transition could be expected to impact recruitment success. There is some evidence of fidelity relating to spawning sites and pre-spawning migration routes, but site-specific investigations are required for confirmation.

Information deficiencies

Although some studies have provided information on the general habitat requirements of burbot more information is needed from the Arctic and tundra regions to adequately understand the habitat requirements of various stages in the life cycle of burbot.

There is a need to determine if the various life history forms (lacustrine, adfluvial and fluvial) of burbot are sufficiently different to represent genetically discrete demes or stocks. If indeed they do, then different strategies may be required to manage them.

Determination of burbot migrations and site fidelity are required. In addition it is important to determine the competitive interaction between burbot and other species since there is a broad diet overlap between burbot and other piscivorous fish species.

Table 14. Habitat and biological characteristics of burbot (*Lota lota*) (adapted from Ford et al. 1995).

Eggs	
Temperature	1.0-7.0 °C
Incubation Temperature	4-7 °C (optimum)
Incubation Time	30-90 days depending upon temperature
Dissolved oxygen	Recommended 6.5 mg/L (interstitial water)
pH	
Current Velocity	<8 cm/s
Substrate	Interstices of gravel, sand and cobble free of silt and debris
Cover	
Larvae and Juveniles	
Temperature	8.0-23.3 °C; optimum for growth 15.6-18.3 °C
Dissolved Oxygen	Recommended >7.75 mg/L; general critierion for freshwater fish calculated by Davis (1975).
Habitat	Rivers, lakes, tributaries
Preferred Depth	Dependent on temperature. Larvae move to deeper waters with the rise in temperature in summer
Preferred Current Velocity	
Substrate	Gravel, rock, cobble
Cover	Rocks, cobble, vegetation, logs – ontogenetic link
Turbidity/TSS	
Primary Food	Aquatic insects
Secondary Food	Fish
Adults	
Temperature	0.6-23.3 °C; optimum for growth 15.6-18.3 °C
Dissolved Oxygen	Recommended >7.25 mg/L; general critierion for freshwater fish calculated by Davis (1975)
Habitat	Rivers, lakes
Preferred Depth	Dependent on temperature; generally deep pools of rivers and hypoliminion of lakes
Preferred Current Velocity	<30 cm/s
Substrate	Gravel, cobble, rock
Cover	Darker profundal waters
Turbidity/TSS	1,000 NTU may hamper feeding; data limited
Primary Food	Fish
Secondary Food	Aquatic insects
Spawning	
Spawning Habitat	Shoals, shores in lakes and rivers
Spawning Substrate	Gravel, sand and cobble free of silt and debris
Nest Construction	None
Spawning Temperature	0.6-1.7 °C
Spawning Depth	0.3-3.0 m
Spawning Current Velocity	<122 cm/s in river mouths with upwelling
Fecundity	45,000-1,000,000 eggs/female
Age at Maturity	2-8 years depending upon geographical location

Table 15. Matrix indicating state of information on the biology and ecology of burbot (*Lota lota*) in the Canadian Arctic.

Topics			**Life Cycle Stages**			
			Eggs	Larvae	Juveniles	Adults
Ecology and Habitat Requirements	Critical Habitat	Summer	n/a			**M**
		Winter	n/a			**L**
	Reproduction	Timing	n/a	n/a	n/a	**M**
		Habitat	n/a	n/a	n/a	**M**
	Rearing and Refugia	Summer	n/a			
		Winter	n/a			
	Food		n/a			**M**
	Water	Depth and Velocity				**M**
		Quality				**L**
Behavior	Feeding		n/a			**L**
	Predation and Predator Avoidance					**L**
	Migration		n/a			**M**
Physiology/Biochemistry	Growth		n/a			**L**
	Function[4]					
Trophic Linkages[1] and Bioenergetics[2]						**L**
Populations[3]			n/a			**M**
Human Impacts						

H = High; M = Medium; L = Low; n/a = not applicable. Blank boxes indicate no information available.
1 = Trophic linkages among or between species; 2 = Bioenergetics (balance between energy intake in the form of food and energy utilization by animals for life-sustaining processes) includes role of external factors such as global warming and pollution; 3 = Populations includes limiting factors and vulnerability to environmental changes; 4 = Ability to adapt/tolerate environmental changes.

Table 16. Matrix indicating state of information on the biology and ecology of burbot (*Lota lota*) in in areas south of 60°N.

Topics			Life Cycle Stages			
			Eggs	Larvae	Juveniles	Adults
Ecology and Habitat Requirements	Critical Habitat	Summer	n/a	L	L	M
		Winter	n/a			L
	Reproduction	Timing	n/a	n/a	n/a	H
		Habitat	n/a	n/a	n/a	M
	Rearing and Refugia	Summer	n/a	L	L	L
		Winter	n/a			
	Food		n/a	L	L	H
	Water	Depth and Velocity	L	L	L	M
		Quality	L	L	L	L
Behavior	Feeding		n/a			L
	Predation and Predator Avoidance					L
	Migration		n/a	L	L	M
Physiology/Biochemistry	Growth		n/a	L	L	H
	Function[4]		L	L	L	L
Trophic Linkages[1] and Bioenergetics[2]			L	L	L	M
Populations[3]			n/a	L	L	M
Human Impacts						L

H = High; M = Medium; L = Low; n/a = not applicable. Blank boxes indicate no information available.
1 = Trophic linkages among or between species; 2 = Bioenergetics (balance between energy intake in the form of food and energy utilization by animals for life-sustaining processes) includes role of external factors such as global warming and pollution; 3 = Populations includes limiting factors and vulnerability to environmental changes; 4 = Ability to adapt/tolerate environmental changes.

LONGNOSE SUCKER *(Catostomus catostomus)*

The habitat and biological characteristics of longnose sucker are summarized in Table 17. Tables 18 and 19 provide a qualitative assessment of the state of knowledge about the biology of longnose sucker in the Arctic and areas south of 60°N respectively.

Distribution

This species is widely distributed in North America: from New England to Labrador in the east, westward through the Laurentian Great Lakes, the northern part of the Mississippi-Missouri River system and in the Columbia River drainage to the Pacific Coast (Morrow 1980). It is common throughout NWT and Nunavut. The species exhibits lacustrine, adfluvial and riverine life history forms (McPhail and Lindsey 1970; Scott and Crossman 1973; Morrow 1980). Longnose sucker is known to hybridize with white sucker, *C. commersoni* (Nelson 1973; Richardson et al. 1994; Dion et al. 1994).

Biology of life history stages

Adults: Adults occur in lakes, rivers and streams throughout their geographic range. The lacustrine individuals use the deep waters in cold, oligotrophic lakes and enter rivers only to spawn (Edwards 1983). There is considerable discrepancy in depths frequented by longnose suckers. According to Scott and Crossman (1973), adult longnose suckers usually inhabit deep waters up to 183 m. Harris (1962) on the other hand has observed them in 1-24 m deep waters in Great Slave Lake. McPhail and Lindsey (1970) have also reported that in Great Slave Lake they are uncommon below 17 m depth.

One study in a Maine reservoir (Moring et al. 1986) has shown that longnose suckers are attracted to submerged pulpwood logs.

In northern locations, such as Great Slave Lake, longnose suckers grow larger and live much longer than those in more southerly locations (Harris 1962; Scott and Crossman 1973). Scott and Crossman (1973) maintain that adults are generally omnivorous, but Morrow (1980) stated that longnose suckers feed almost entirely on benthic material. In Great Slave Lake, they are reported to feed mainly on amphipods, chironomids, midge larvae, caddisfly larvae and sphaeriid clams (Scott and Crossman 1973).

Adults have been reported to prefer water temperatures between 10 and 15 °C (Brown and Graham 1954, cited in Edwards 1983). In laboratory experiments that assessed the thermal tolerance of adults, those caught in waters at 14.4 °C died at 28.3 °C (Cooper and Fuller 1945, cited in Edwards 1983). The upper temperature at which 50% of longnose suckers from lakes in the Okanagan Valley died in 24 h was estimated to be about 27 °C (acclimation temperatures between 11.5 °C and 14 °C (Black 1953)).

Larvae and juveniles: Hatched fry are about 8 mm in length and stay in gravel substrates for 1-2 weeks (Geen et al. 1966; McPhail and Lindsey 1970; Scott and Crossman 1973; Morrow 1980). Young frequent shallow areas of lakes and are often found associated with vegetation and sandy substrates (Edwards 1983). According to Morrow (1980), nocturnal downstream movement of fry in some areas of Alaska begins as soon as fry (alevins) emerge from the gravel, but in interior Alaska some stay in the streams all summer long.

Eggs: Eggs are about 3 mm in diameter, yellow in color, adhesive and are broadcast over sandy and gravelly riverine substrates in 10-60 cm deep water with currents between 30 and 45 cm/s (Geen et al. 1966). Eggs sink to the bottom where they settle in crevices in the substrate and hatch in 1-2 weeks, depending upon temperature (Morrow 1980).

Ecology of life history stages

Habitat use: Longnose sucker is essentially a deep water species which prefers cold, oligotrophic habitats (Scott and Crossman 1973). The species is most abundant in cold, oligotrophic lakes that are 30-40 m deep that have little and steeply-sloped littoral areas (Walton 1980). These lakes have <10-20 mg/L total dissolved solids (TDS) and Secchi disk readings between 4 and 13 m. Dissolved oxygen concentrations in lakes abundant with longnose suckers have ranged from 5.6-10 mg/L (Rawson 1942; Johnson 1971, cited in Edwards 1983).

For most of their lives, longnose suckers are generally benthic in habit. They are typically stream spawners in riffled areas with cobble and gravel substrate (Geen et al. 1966; McPhail and Lindsey 1970; Scott and Crossman 1973; Morrow 1980). However, they are also known to use wave-swept, rocky shorelines of lakes in 15-30 cm deep waters (Geen et al. 1966; Walton 1980), and they have also been known to use sandy substrates (Geen et al. 1966; Morrow 1980). They are essentially benthic feeders as adults. YOY occupy and feed in habitats which have "quiet" waters with vegetation, boulders or rubble (Edwards 1983).

Spawning: Spawning takes place in spring (April to June) after ice melt (McPhail and Lindsey 1970; Scott and Crossman 1973) and can commence as late as July in northerly locations (Morrow 1980). Upstream spawning migrations have been documented to occur between noon and midnight with the greatest movement in evening hours; spawning has been observed within 0600-0900 h (Harris 1962; Geen et al. 1966; Scott and Crossman 1973). Spawning migrations in Frye Creek (an inlet stream of Sixteenmile Lake, BC) began when daily maximum stream temperatures rose above 5 °C, but the greatest intensity was observed above 10 °C. Spawning has been documented to occur every year in the Northwest Territories (Tripp and McCart 1974).

In Great Slave Lake, the age of spawning longnose suckers ranged between 9 and 15 years (Carl et al. 1967, cited in Richardson et al. 2001).

Longnose suckers spawn primarily in rivers and streams but may also spawn in shallow areas of lakes (Harris 1962; Geen et al. 1966; Scott and Crossman 1973; Morrow 1980). The river-spawning fish move from lakes into inlet streams with gravel ranging in size from 0.5 to 10 cm in diameter in shallow waters 10-60 cm deep, and with water currents between 30 and 45 cm/s (Geen et al. 1966).

The lake populations tend to spawn in depths between 15 and 30 cm along rocky wave-swept shorelines with gravel and sand substrates (Geen et al. 1966; Scott and Crossman 1973).

A single male can mate with several females at the same time and females are reported to spawn many times within an hour. Over 60,000 eggs may be produced by a single female (Harris 1962; Geen et al. 1966). Tripp and McCart (1974) found a significant positive correlation between fecundity and fork length of longnose suckers in the Donnelly River system, Northwest Territories. No nests are built and eggs are broadcast and sink to the substrate (Morrow 1980).

The lacustrine populations which move out of the lake to spawn return to the lake a few days after spawning is completed, but the riverine populations may stay at spawning sites or in that general area for much of the summer (Morrow 1980).

Spawning mortality of adults has been reported to be 10 to 30% in Sixteenmile Lake, British Columbia (Geen et al. 1966). While some fish have been reported to spawn at two- and three-year intervals, the majority spawn in two or even three consecutive years (Geen et al. 1966). In Gouin Reservoir System (Quebec), Richardson et al. (1994) also provided evidence that some longnose suckers skipped a year to spawn 2 years later.

Fidelity to spawning streams has been demonstrated in British Columbia (Geen et al. 1966) and in the Donnelly River system in the Northwest Territories (Tripp and McCart 1974).

In many watersheds of Canada both longnose and white suckers undergo simultaneous spawning migrations raising the risk of hybridization (Nelson 1968, 1973; Dauble and Buschbom 1981). Interspecific mating between these two species (Richarson et al. 1994) and occurrence of hybrids (Nelson 1973) has been observed.

Rearing: Larvae are about 8 mm long at hatching and stay in the substrate for 1-2 weeks (Geen et al. 1966; McPhail and Lindsey 1970; Scott and Crossman 1973; Morrow 1980). Earlier emergence is considered to be hampered by the possession of large yolk sacs which affect the ability to swim (Walton 1980). At emergence, the fry are about 12 mm long with shrunken yolk sacs (Walton 1980). After emergence, young frequent shallow areas of lakes, often in association with vegetation and sandy substrates (Richardson et al. 2001). In streams, the newly-hatched larvae remain in the spawning substrate (gravel) for about two weeks before emerging and moving downstream (Geen et al. 1966; Scott and Crossman 1973; Tripp and McCart 1974; Walton 1980). According to Walton (1980), during downstream movement, fry are most commonly found in relatively fast-flowing water at or near the surface. Chang-Kue and Cameron (1980) reported that fry moved out from the tributary streams of the Great Bear River (Northwest Territories) into the main channel by mid July. These authors found the YOY of longnose sucker along the Great Bear River during August and September.

In interior Alaska some young stay in the streams all summer (Morrow 1980). Downstream movement of fry in Sixteenmile Lake, BC has been reported to occur between 2300 and 0100 h, which is generally the period of lowest nocturnal illumination at that time of the year and location (Geen et al. 1966).

Fry are planktivorous and make the transition to feed on larger prey as they grow (Barton and Bidgood 1980).

Food supply: Longnose sucker almost exclusively feed on benthic organisms (Morrow 1980). In Paine Lake (Alberta), however, they have been reported to feed mostly on *Daphnia* and other cladocerans, particularly in July and August (Barton 1980; Barton and Bidgood 1980). In this lake, cladocerans (mostly *Daphnia*) formed 74.2% of the total food volume of the longnose sucker diet, 33.7% of white sucker diet, and 18% of rainbow trout diet (Barton and Bidgood 1980). In earlier studies, other authors also reported cladocerans as major food items for longnose suckers (Brown and Graham 1954; Beamish 1974; Lalancette 1977).

Longnose sucker feed on the benthos by slowly moving over the bottom substrate with their mouths touching and sucking in the food (Morrow 1980). Major food items of stream-dwelling adults include algae and other benthic vegetation, diptera, ephemeroptera, coleoptera, trichoptera, spiders and mollusks, while the lake-dwelling adults mainly rely on various crustaceans, particularly cladocerans, amphipods and insect larvae and nymphs (Morrow 1980). They are also reported to feed on fish eggs, including trout and whitefish eggs (Morrow 1980). Intense predation on lake whitefish eggs by longnose suckers has been observed by Nester and Poe (1984) in northwestern Lake Huron.

Larvae of white sucker (*Catostomus commersoni*) a species closely related to longnose sucker are reported to feed on littoral planktonic organisms (Siefert 1972). They commence feeding when 12 to 13 mm long and the yolk sac is not completely absorbed (the absorption is completed when the fish reaches about 14 mm in length). They exhibit little preference for specific food organisms, but rotifers belonging to several genera contribute an important portion of early diet.

Refugia: Longnose suckers are considered to over winter in deep waters of lakes and large rivers (Morrow 1980).

Behavior

Feeding and predator avoidance: As benthic feeders, they compete for food with all other bottom feeders except those from whom they are spatially separated in deeper waters (Scott and Crossman 1973). They are reported to aggressively prey on the eggs of lake whitefish and other salmonids (Nester and Poe 1984).

In Paine Lake, Alberta, competition for food was observed in 1977 between longnose suckers and white suckers and with introduced rainbow trout (Barton and Bidgood 1980). The extent of piscivore predation on longnose sucker fry has not been documented.

In a laboratory study Symons (1976) observed no apparent competitive advantage or disadvantage between white suckers (*Catostomus commersonii*) and three other competitors (Atlantic salmon, *Salmo salar;* common shiner, *Notropis cornutus;* and blacknose dace, *Rhinichthys atratulus*). All these three competitors preferred habitats that simulated their natural ones.

Migration: With the exception of movement to and from spawning sites, no other definitive migrations have been reported for longnose sucker. During summer in Alaska, random movement has been observed in rivers, some moving upstream and others moving downstream (Morrow 1980). Longnose suckers generally spend their juvenile years in lentic waters and return to streams for spawning as adults, as previously mentioned (Walton 1980). Olson and Scidmore (1963) provided evidence that white suckers have an affinity to a particular spawning site and repeat spawners return to a given stream from all parts of a lake. Geen et al. (1966) also provided evidence of homing behavior of both white and longnose suckers to the same stream for spawning in successive years in Sixteenmile Lake, BC. There is no similar information with respect to Arctic tundra lakes.

In a study of the movements of radio-tagged Arctic grayling, lake trout and longnose sucker between Kodiak Lake and Lac de Gras, Northwest Territories, Low (2002) concluded that "Arctic grayling and longnose sucker are capable of moving between all lakes and streams between Lac de Gras and Kodiak Lake (the site of BHP Billiton's Ekati diamond mine) during springtime."

Physiology and biochemistry

Growth, fitness and survival: Growth has been reported to vary substantially among locations (Walton 1980), "apparently correlated with food supply rather than temperature" (Morrow 1980). This statement, however, appears to be highly speculative because the literature suggests various causes for the variation in growth rates observed among populations of longnose suckers. Tripp and McCart (1974) found considerable discrepancy between the results of ageing by scales and otoliths and suggested that this should be taken into consideration when comparing growth rates in different populations. Underlying causes of slow growth in some populations have been attributed to limited food and space, overpopulation (Beamish and Crossman 1977), and lake acidification (Beamish and Harvey 1972). Some evidence of genetic differences between dwarf and large forms has been provided by Beamish and Tsuyuki (1971).

In Great Slave Lake, YOY may reach 8 cm fork length by the end of August. In a more southerly location (Yellowstone Lake, Wyoming) the YOY are reported to be less than half that length (Harris 1962; Tripp and McCart 1974; Morrow 1980).

In Great Slave Lake, longnose suckers along the south shore have been reported to grow faster than suckers in the more northerly areas of the lake (Rawson 1951; Harris 1962). However, it is possible that discharges rich in organic material from the Slave, Buffalo and Hay Rivers may have provided more favorable conditions for growth as compared to the essentially more oligotrophic conditions in the northern areas of the lake (Harris 1962). Variations in growth rate have been recorded; within a population the largest fish of a given age may be twice as long as the smallest individual (Geen et al. 1966; Morrow 1980).

Age at maturity: Age at maturity varies considerably and increases with latitude (Harris 1962; Morrow 1980); for example, 2+ for males and 3+ for females (Colorado), 4+ or 5+ (Yellowstone Lake, Wyoming), and 9+ or 10+ (Great Slave Lake, NWT).

Trophic linkages

In a study of competitive feeding habits of rainbow trout, white and longnose suckers in Paine Lake, Alberta, Barton and Bidgood (1980) demonstrated competition for food between the introduced rainbow trout and resident white and longnose suckers. While the food for longnose suckers was primarily *Daphnia*, rainbow trout were opportunistic feeders, utilizing pelagic and terrestrial insects when available and feeding primarily on *Daphnia* and chironomid larvae when these pelagic and terrestrial insects were not present. Feeding selectivity for *Daphnia* over smaller zooplankton was exhibited by all three species.

Longnose suckers are known to feed aggressively on fish eggs (Morrow 1980). Nester and Poe (1984) observed intense predation by longnose suckers on eggs of lake whitefish (*Coregonus clupeaformis*). They attributed this predation to the spawning habits of coregonines as broadcast spawners, depositing their eggs over less-protective, gravelly shallows, unlike members of *Salmoninae* (salmons, trouts, and charrs) that deposit their eggs over or in bottom substrates which provide protection from most fish predators.

Critical factors for survival

Harris (1962) found that the growth rate of longnose suckers was faster in organic-rich waters of southern Great Slave Lake than the northern part of the lake which more oligotrophic. Similarly, Ryan (1980, cited in Edwards 1983) observed that growth of longnose sucker was lowest in rapid waters below a waterfall and fastest in the slow-moving waters downstream, which were organically rich. In both these studies the variation in growth rate was attributed to the availability of food.

Adequate dissolved oxygen and suitable depth are critical in over-wintering areas with ice cover (see preferred depth for adults).

Information deficiencies

The list of information deficiencies is similar to other species reviewed since little is known about the biological and ecological characteristics of this species in Arctic tundra lakes and streams. Of particular importance is information on interactions with other species (predation, competition for food and space), extent of fidelity to spawning grounds, feeding areas and substrates, and their ontogenetic habitat tolerance limits to environmental factors such as hypoxic conditions, pH, turbidity, and suspended solids.

Table 17. Habitat and biological characteristics of longnose sucker, *Catostomus catostomus*.

Eggs	
Temperature	
Incubation Temperature	10-15 °C
Incubation Time	1-2 weeks; 7 days at 17 °C, 14 days at 12 °C
Dissolved oxygen	
pH	
Current Velocity	30-45 cm/s
Substrate	Sand and gravel
Larvae and Juveniles	
Temperature	
Dissolved Oxygen	
Habitat	Shallow and often vegetated and sandy areas
Preferred Depth	
Preferred Current Velocity	
Substrate	Sand, gravel
Cover	Vegetation
Turbidity/TSS	
Primary Food	Plankton
Secondary food	Aquatic insects
Adults	
Temperature	Prefer temperatures between 10° and 15 °C. Upper lethal temperature 26-27 °C
Dissolved Oxygen	5.6-10 mg/L
Habitat	Lakes, streams and rivers with clear waters
Preferred Depth	Primarily bottom-dwelling; most abundant in oligotrophic lakes 30-40 m deep. Also reported from shallower waters
Preferred Current Velocity	
Substrate	
Cover	
Turbidity/TSS	
Primary Food	Benthic organisms
Secondary Food	Fish and fish eggs (e.g. lake whitefish eggs)
Spawning	
Spawning Habitat	Rivers and streams, shallow areas of lakes
Spawning Substrate	Large rocks 10-50 cm diameter, or sand and gravel
Nest Construction	None
Spawning Temperature	5-10 °C
Spawning Depth	10-60 cm
Spawning Current Velocity	30-45 cm/s
Fecundity	17,000 to 60,000 eggs/female
Age at Maturity	2-10 y depending on latitude; older at higher latitudes

Table 18. Matrix indicating state of information on the biology and ecology of longnose sucker (*Catostomus catostomus*) in the Canadian Arctic.

Topics			**Life Cycle Stages**			
			Eggs	Larvae	Juveniles	Adults
Ecology and Habitat Requirements	Critical Habitat	Summer	n/a			L
		Winter	n/a			
	Reproduction	Timing	n/a	n/a	n/a	M
		Habitat	n/a	n/a	n/a	M
	Rearing and Refugia	Summer	n/a			
		Winter	n/a			
	Food		n/a			M
	Water	Depth and Velocity				M
		Quality				
Behavior	Feeding		n/a			M
	Predation and Predator Avoidance					
	Migration		n/a			M
Physiology/Bio-chemistry	Growth		n/a			M
	Function[4]					
Trophic Linkages[1] and Bioenergetics[2]						L
Populations[3]			n/a			L
Human Impacts						

H = High; M = Medium; L = Low; n/a = not applicable. Blank boxes indicate no information available.
1 = Trophic linkages among or between species; 2 = Bioenergetics (balance between energy intake in the form of food and energy utilization by animals for life-sustaining processes) includes role of external factors such as global warming and pollution; 3 = Populations includes limiting factors and vulnerability to environmental changes; 4 = Ability to adapt/tolerate environmental changes.

Table 19. Matrix indicating state of information on the biology and ecology of longnose sucker (*Catostomus catostomus*) in areas south of 60°N.

Topics			Life Cycle Stages			
			Eggs	Larvae	Juveniles	Adults
Ecology and Habitat Requirements	Critical Habitat	Summer	n/a			M
		Winter	n/a			
	Reproduction	Timing	n/a	n/a	n/a	M
		Habitat	n/a	n/a	n/a	M
	Rearing and Refugia	Summer	n/a	L	L	L
		Winter	n/a			
	Food		n/a	L	L	M
	Water	Depth and Velocity	L	L	L	M
		Quality				L
Behavior	Feeding		n/a			M
	Predation and Predator Avoidance					M
	Migration		n/a			M
Physiology/Biochemistry	Growth		n/a			M
	Function[4]					
Trophic Linkages[1] and Bioenergetics[2]						L
Populations[3]			n/a			L
Human Impacts						

H = High; M = Medium; L = Low; n/a = not applicable. Blank boxes indicate no information available.
1 = Trophic linkages among or between species; 2 = Bioenergetics (balance between energy intake in the form of food and energy utilization by animals for life-sustaining processes) includes role of external factors such as global warming and pollution; 3 = Populations includes limiting factors and vulnerability to environmental changes; 4 = Ability to adapt/tolerate environmental changes.

SLIMY SCULPIN (*Cottus cognatus*)

The habitat and biological characteristics of slimy sculpin are summarized in Table 20. Tables 21 and 22 provide a qualitative assessment of the state of knowledge about the biology of slimy sculpin in the Arctic and areas south of 60°N respectively.

Distribution

This species is distributed from northern North America to extreme northeastern Siberia (Scott and Crossman 1973). In North America, it occurs in more northerly waters than related *Cottus* species such as *C. bairdii*. It commonly occurs from Virginia, Labrador, and Ungava in the east to Alaska and St. Lawrence Islands in the Bering Sea. It occurs throughout the Northwest Territories and Nunavut in rivers, streams and lakes, but it is reported to be absent from the Arctic islands and the mainstem of the Mackenzie River (McPhail and Lindsey 1970).

In northern freshwaters of North America, slimy sculpin is the most widely-distributed member of its genus (*Cottus*). In some Arctic drainages, such as the Bering, Chukchi, and Beaufort, slimy sculpin is virtually the only freshwater sculpin present (McPhail and Lindsey 1970; Scott and Crossman 1973). In the Great Lakes, slimy sculpin are abundant and are considered the most numerous fish in the offshore benthos of Lake Ontario (Scott and Crossman 1973; Christie and Thomas 1981, cited in Brandt 1986*a,* 1986*b*).

Slimy sculpin inhabit cold, well-oxygenated waters in lakes and streams, usually in association with various salmonids (McPhail and Lindsey 1970; Craig and Wells 1976; McDonald et al. 1982; Hughes and Penden 1984). The species is known to exhibit both lacustrine and riverine life history types (McPhail and Lindsey 1970; Scott and Crossman 1973; Hughes and Penden 1984).

Slimy sculpin shares its habitat with other freshwater cottids (such as the deepwater sculpin, *Myoxocephalus thompsonii*; and spoonhead sculpin, *Cottus ricei*) (Sonnichsen 1981; Mohr 1984, 1985; Selgeby 1988) and ecologically-similar species of riffle communities such as the darter, *Etheostoma olmstedi* (Gray and Stauffer 1999).

Biology of life history stages

Adults: The habitat of slimy sculpin adults in the open-water period in the Arctic is clear streams with gravel (Tripp et al. 1981). In southern locations such as Pennsylvania, New York and Wisconsin, adults can be found under cobble, boulders, bedrock, rubble, and sand substrates in 10-30 cm deep waters within a velocity range of 0.04 to 0.46 m/s (Becker 1983; Johnson et al. 1992; Gray and Stauffer 1999; Baldigo and Lawrence 2001). Lacustrine sculpins have been found at depths ranging between 0.5 to 210 m and are usually associated with gravel and rocky substrates (Selgeby 1988).

Eggs: Sculpin eggs are adhesive and are laid on the ceiling of a nest (under rock, ledge or submerged tree root) (Scott and Crossman 1973).

Larvae and juveniles: Slimy sculpin larvae are benthic and have no planktonic phase (Heard 1966). After hatching, fry fall to the bottom of the nest where they remain until the yolk sac is fully absorbed (3 to 6 days). Afterwards young sculpins start to leave the nest (Koster 1936, cited in Evans et al. 2002). Young of the year have been found under cobble and rubble substrate in 13-20 cm of water and in water velocities between 0.06 and 0.56 m/s (Gray and Stauffer 1999). Juveniles are found on similar substrates but in deeper waters (10-30 cm) with stronger currents ranging between 5 and 40 cm/s (Johnson et al. 1992). In lakes, young are commonly found over gravel and sandy substrates in waters 0.5-1.5 m deep (Mohr 1984). As they grow, they gradually move from shallow waters of the lake to deeper waters (Mohr 1985; Brandt 1986*c*).

Ecology of life history stages

Habitat use: The habitat for slimy sculpin typically consists of gravel and rocks which they occupy during the day (probably to escape predation). In the Great Lakes, their distribution ranges from nearshore (typically >10 m depth) to depths of over 150 m in Lake Ontario (Brandt 1986*c*) and 210 m in Lake Superior (Selgeby 1988). But in these lakes, they are more common at less than 100 m depth (Brandt 1986*c*; Selgeby 1988). With the exception of probable movement into shallow waters for spawning, slimy sculpin apparently do not migrate far and, like other benthic, insectivorous fishes such as darters, they are relatively sedentary with small home ranges (Morrow 1980; Gray et al. 2004).

In Lake Ontario, evidence has been provided for ontogenetic shifts in habitat, diet and diel-feeding periodicity as slimy sculpins mature (Brandt 1986*c*). Sculpins less than 50 mm total length generally inhabited water shallower than 60 m but larger fish dominated catches at 75 m; sculpins at 35 m fed mainly at night but those feeding at 75 m depth showed no diel-feeding periodicity. At an intermediate depth of 60 m, more sculpins were caught at night than during day. The day and night catches did not differ significantly at 75 m depth. These data are suggestive of behavioral adaptations to avoid predation.

Environmental factors considered to influence the distribution of sculpins in a lake include food availability, habitat, water temperature, oxygen concentration, territory size, and light penetration (Van Vliet 1964, cited in Mohr 1984).

Spawning: Slimy sculpin spawn in the spring, shortly after ice breakup when water temperatures are between 4.5 and 10 °C (Morrow 1980). This situation may occur as early as late March in the southern part of the range of slimy sculpin or May/June in the north (Scott and Crossman 1973; Morrow 1980).

In Alaskan streams, spawning takes place in late May and early June when water temperature is about 3.5 °C (Craig and Wells 1976). In other locations such as the Montreal River, spawning is reported to occur in early May when water temperature is around 8 °C (Scott and Crossman 1973; Lee et al. 1980). Males select a nesting site under a rock or ledge and court the female into the nest. After egg deposition the female leaves the nest and the male guards it (Scott and Crossman 1973; Lee et al. 1980). Slimy sculpins are usually monogamous but males may spawn with one or more females (Mousseau and Collins 1987). Lacustrine populations of slimy sculpins spawn in shallow waters <1.5 m deep (Morrow 1980; Mohr 1985; Mousseau and Collins 1987).

Fecundity ranges between 42 and 1,420 eggs; the number of eggs increases with the size of the female (Morrow 1980). A comparative study by Foltz (1976) in the Great Lakes suggested that lake-spawning populations of slimy sculpin may be more fecund than cottids living in streams and small lakes. Furthermore, Foltz (1976) also found that "for the slimy sculpin, fecundity expressed as a linear function of somatic weight produced a good fit."

As in many other fish, temperature has been considered to be the most important determinant of spawning time (Mohr 1984; Van Vliet 1964, cited in Craig and Wells 1976). Eggs may require four weeks to hatch at a temperature around 8 °C (Scott and Crossman 1973). In Alaska, fry have been found in streams in late June (Craig and Wells 1976).

Rearing: After hatching, the fry fall to the bottom of the nest where they remain for 3-4 days or until the yolk sac is consumed. Young of the year have been found under cobble and rubble substrates in Pennsylvania streams in 13-22 cm deep water and with a velocity of 0.06 to 0.56 m/s (Gray and Stauffer 1999). Johnson et al. (1992) reported that in a Pennsylvania River (Straight Run River), YOY were most common at depths of 5-25 cm with water velocities <20 cm/s. Juvenile sculpins in the same river occupied deeper waters (10-30 cm) and areas with stronger currents (5-40 cm/s).

Food supply: Sculpins are essentially benthivores and adults of both river and lake populations feed mainly on stream benthos (Craig and Wells 1976). However, the relative importance of taxa eaten by slimy sculpin varies considerably. For example, in Lake Superior, amphipods (*Pontoporeia affinis*) are reported to be the dominant food (93% of total biomass consumed by slimy sculpins) in all months of the year (Selgeby 1988). Brandt (1986*b*) reported similar feeding patterns for slimy sculpin in Lake Ontario. The diet of slimy sculpin in the Chena River (Alaska) mainly consisted of chironomids (Sonnichsen 1981). The food of slimy sculpin in the Chandalar River (a tributary of the Yukon River in Alaska) consisted almost exclusively of the larvae of stream-dwelling insects, mainly chironomids but also other dipteran larvae, plecopteran and ephemeropteran nymphs (Craig and Wells 1976).

In the Laurentian Great Lakes, slimy sculpins and other species of the genus *Cottus* are known to prey on lake trout eggs and larvae (Hudson et al. 1995). Similarly, Foote and Brown (1998) showed that in Iliamna Lake (Alaska), sculpins of genus *Cottus* (*C. aleuticus* and *C. cognatus*) "actively move to specific spawning beaches and that the initiation of their movements precedes the start of spawning." These authors also observed that "sculpin predation on sockeye eggs is positively dependent on sculpin size and on the state of eggs (fresh versus hardened), with the largest sculpins able to consume nearly 50 fresh eggs at a single feeding and 130 over a 7-day period."

In Lake Ontario, Fitzsimons et al. (2002) estimated the abundance of sculpins (*Cottus* spp.) and crayfish (*Orconectes* sp.) to assess predation on lake trout eggs by these predators and concluded that "egg consumption by sculpins could be a significant source of mortality at some Lake Ontario reefs where egg abundance is low (<100/m^2). Some other authors (Biga et al. 1998; Hudson et al. 1995; Chotkowski and Marsden 1999) have come to similar conclusions and deduced that predation by sculpins on salmonid eggs could be critical for the recruitment of the latter.

Since in many Arctic tundra lakes, lake trout and slimy sculpin are sympatric (Martin 2001), it would be important to understand the predation by sculpin on lake trout eggs.

Migration and refugia: Slimy sculpin are essentially a sedentary benthic species. With the probable exception of inshore spawning migrations, slimy sculpin show limited movement (Morrow 1980; Gray et al. 2004).

Behavior

Feeding and predator avoidance: Feeding activity for slimy sculpin is highest at night. In Lake Superior, the average stomach content of slimy sculpins was greatest at 0630 and declined linearly thereafter through an 8-hour daylight period (Selgeby 1988). This linear decline in food consumed and the general deteriorating condition of food organisms in stomachs suggested that few fish continued to feed during the day.

Evidence has been provided to suggest that slimy sculpins are not generalists in their feeding behavior. For example, in the Chena River in Alaska, they show preferences for large mayflies and chironomids (Sonnichsen 1981). In such situations, preferred food availability may become a critical factor for survival.

Predation of sculpins on salmonid eggs has been reported by a number of authors (Hudson et al. 1995; Foote and Brown 1998; Biga et al. 1998; Chotkowski and Marsden 1999; Fitzsimons et al. 2002). Evidence has been provided to conclude that predation of sculpins on salmonid eggs is guided by chemical cues emanating from the eggs rather than visual cues. In a series of two-choice maze experiments, Dittman et al. (1998) showed that "sculpins preferred egg wash (from sockeye salmon eggs) to lake water but demonstrated no attraction to ovarian fluid versus lake water, suggesting that the attractive substances are derived directly from egg material." These authors also demonstrated this exclusive reliance on chemosensory cues to locate salmon eggs by slimy sculpins in field experiments in Iliama Lake (Alaska), "where substantial numbers of free, unmanipulated sculpins in sockeye spawning areas entered minnow traps baited with eggs that could be smelled but not seen." Mirza and Chivers (2002) demonstrated a similar chemosensory response of slimy sculpins towards brook charr (*Salvelinus fontinalis*) eggs in Maine (USA).

The role of both visual and chemical cues has been demonstrated in "threat-sensitive predator avoidance" behavior of slimy sculpins by Chivers et al. (2001). When only chemical cues from the predatory brook trout were presented to slimy sculpins they avoided brook trout regardless of size. In field experiments, however, with caged, small (average standard length 59.4 mm) and large brook trout (average standard length 130.7 mm), sculpins avoided only the large trout, suggesting that "chemical cues function to warn the sculpin that the predator is in the vicinity, but visual cues are needed in order to accurately assess the risk posed by the predator."

Migration: In Lake Ontario, evidence has been provided with respect to ontogenetic shifts in slimy sculpin habitat (Brandt 1986*c*). Brandt's data suggest that young sculpins migrate from shallow water habitat to deeper water habitat as they mature. This ontogenetic shift in habitat also provides the sculpins with the opportunity to feed continuously in deeper waters, as opposed to restricting their nocturnal feeding in shallow waters as when they were young. This behavior is probably related to avoidance of predators (Brandt 1986*c*).

In a study of the depth distribution of slimy sculpin in a small lake in northwestern Ontario, Mohr (1985) found that the distribution of slimy sculpin changed both seasonally and diurnally. In the spring and fall, when the study lake water temperatures ranged from 5 to 13 °C and dissolved oxygen concentrations ranged from 1.5 to 9.5 mg/L, slimy sculpins were seen at all depths in the lake. When the lake was thermally stratified, sculpins congregated in the metalimnion and showed a distinct diurnal distribution: during the day they were seen in the upper regions of metalimnion (temperature 10-20 °C, dissolved oxygen 8-11 mg/L) and at night they were seen congregating in the lower part of the hypolimnion (temperature 7-10 °C, congregating in waters with dissolved oxygen 8-10 mg/L).

Physiology and biochemistry

Growth, fitness and survival: Decreased growth and increased longevity is considered common in fish populations from higher latitudes (Rounsfell 1975, cited in McDonald et al. 1982; Craig and Wells 1976). For example, in the Chena River, Alaska, slimy sculpins take almost 5 years to attain the length reached in 2 years by those in Minnesota (Petrosky and Waters 1975; Sonnichsen 1981). In the Arctic, however, slimy sculpins live longer than those in Minnesota: the oldest in the Chena River (Alaska) being 7 years old as opposed to the oldest found in Minnesota at only 5 years old (Sonnichsen 1981). Other authors have reported similar differences in growth (Craig and Wells 1976). McDonald and Hershey (1992) have observed shifts in abundance and growth of slimy sculpins in response to changes in the predator population in an Arctic lake in Alaska.

Selgeby (1988) reported that in Lake Superior, increment in length was largest in the first year of life of the slimy sculpin and growth in second and third years was reduced to only about 60% of that attained during the first year. However, weight increment was smallest in the first year, but increased in each succeeding year.

Age at maturity: Age at maturity varies considerably in slimy sculpin populations. For example, in Valley Creek, Minnesota, a mature female was found at age 1, but most (both sexes) matured at age 2. In the Arctic, slimy sculpins grow much slower and mature at ages 3 to 4 (Craig and Wells 1976). Some other studies have shown that age at maturity in slimy sculpins is a function of size with 70 mm total length an approximate minimum (Craig and Wells 1976; Petrosky and Waters 1975; Van Vliet 1964). While this may be true in some areas, evidence in other studies has shown that this cannot be applied as a general rule and it is likely that several factors (such as latitudinal differences) can influence age at maturity (McDonald et al. 1982).

Tolerance and resistance: In laboratory studies, slimy sculpins acclimated at 20 °C, had an upper tolerance limit of 25 °C and preferred temperature of 13 °C (Symons et al. 1976).

For slimy sculpin acclimated to a constant temperature ranging from 5 to 20 °C, the critical thermal maximum varied from 23.5 °C for those acclimated at 5 to 29.4 °C for fish acclimated at 20 °C. The incipient upper lethal temperatures ranged from 18.5 to 23.5 °C, with the ultimate upper lethal temperature about 26.5 °C (Otto and Rice 1977). These authors also found that sculpins were capable of recognizing and responding to high and potentially harmful temperature: the avoidance temperature ranged from 15.2 °C for sculpins acclimated at 5 to 21.5 °C for sculpins acclimated at 15 °C.

Slimy sculpins are reported to avoid oxygen concentrations of ≤2 mg/L (Bond 1963, cited in Mohr 1984) and prefer highly-oxygenated, riffled areas in streams (Gray and Stauffer 1999).

Trophic linkages

Very little is known about the trophic interactions of slimy sculpin with other species. Slimy sculpin is an ecologically-important fish species in lakes and streams because of its role as both as prey and as predator, and given that it competes for food and space with commercially-important species such as Arctic grayling and lake trout.

Slimy sculpin share their habitat with a variety of fish species, included among them are other sculpin species of the genus *Cottus* (Brown 1991), the deepwater sculpin (*Myoxocepalus quadricornis thompsonii*) and salmonids (McPhail and Lindsey 1970; Scott and Crossman 1973). In lakes such as Great Bear and Keller (NWT) slimy sculpins are usually found in rocky areas (<10 m deep) that are subjected to some current or wind action (McPhail and Lindsey 1970). In both of these lakes some slimy sculpins were eaten by lake trout. In the Great Lakes such as Lake Superior, slimy sculpins are reported to be abundant at depths greater than 80 m (Dryer 1966, cited in McPhail and Lindsey 1970) and are an important component of fish communities on reefs and adjacent nursery areas. As such, slimy sculpins overlap with age 0 lake trout and the two species exhibit important trophic interactions, including predation on each other's eggs and larvae, and competition for food resources (Hudson et al. 1995). On the basis of their study of predator-prey relations and competition for food between age 0 lake trout and slimy sculpins in Lake Superior, these authors concluded that on spawning reefs with high sculpin to lake trout ratios, coexistence of the two species may be a "bottleneck" for age 0 lake trout survival. Biga et al. (1998) studied the effect of substrate size on lake trout egg predation by mottled sculpin (*Cottus bairdii*) and showed that substrate size can minimize predation pressure on lake trout eggs by excluding larger sculpins from interstitial spaces.

Critical factors for survival

Availability of preferred habitats and availability of appropriate-size food items are critical factors for survival.

Information deficiencies

Factors governing the production of slimy sculpins are important to determine as they are an important forage fish for lake trout. Other requirements are for information on trophic interactions (especially predation) in Arctic tundra lakes, and the impact of habitat deterioration as a result of changes in water quality and other habitat characteristics.

Table 20. Habitat and biological characteristics of slimy sculpin, *Cottus cognatus*.

Eggs	
Temperature	
Incubation Temperature	
Incubation Time	4 weeks at temperatures around 8 °C
Dissolved oxygen	
pH	
Current Velocity	
Substrate	
Cover	
Larvae and Juveniles	
Temperature	
Dissolved Oxygen	
Habitat	Lakes, rivers and streams
Preferred Depth	Larvae in 5-25 cm water in a Pennsylvania river; juveniles in 10-30 cm water
Preferred Current Velocity	Larvae in <20 cm/s current in a Pennsylvania river; juveniles in 5-40 cm/s currents
Substrate	Cobble, gravel and rubble
Cover	Rocks
Turbidity/TSS	
Primary Food	
Secondary Food	
Adults	
Temperature	Preferred 13 °C, avoidance temperature ranges 15.2 to 21.5 °C; upper tolerance limit 25 °C
Dissolved Oxygen	8-10 mg/L; avoid levels ≤2 mg/L
Habitat	Clear streams and lakes
Preferred Depth	10-30 cm; lacustrine sculpins at depths between 0.5-210 m
Preferred current velocity	0.04 to 0.46 m/s
Substrate	Gravel and rocks
Cover	Boulders, rocks
Turbidity/TSS	
Primary Food	Benthivorous, preferred taxa vary considerably
Secondary Food	Fish eggs
Spawning	
Spawning Habitat	Shallow water in lakes and streams
Spawning Substrate	Rocks, cobble, gravel
Nest construction	Under rocks
Spawning Temperature	4.5-10 °C
Spawning Depth	<1.5 m
Spawning Current Velocity	<20 cm/s
Fecundity	42-1,420
Age-at-Maturity	Varies considerably; 3-4 years in Arctic; in southern areas as young as age 1

Table 21. Matrix indicating state of information on the biology and ecology of slimy sculpin (*Cottus cognatus*) in the Canadian Arctic.

Topics			Life Cycle Stages			
			Eggs	Larvae	Juveniles	Adults
Ecology and Habitat Requirements	Critical Habitat	Summer	n/a	L	L	M
		Winter	n/a			
	Reproduction	Timing	n/a	n/a	n/a	M
		Habitat	n/a	n/a	n/a	M
	Rearing and Refugia	Summer	n/a			L
		Winter	n/a			
	Food		n/a			L
	Water	Depth and Velocity				L
		Quality				L
Behavior	Feeding		n/a			
	Predation and Predator Avoidance					L
	Migration		n/a			L
Physiology/Biochemistry	Growth		n/a			L
	Function[4]					
Trophic Linkages[1] and Bioenergetics[2]						L
Populations[3]			n/a			L
Human Impacts						

H = High; M = Medium; L = Low; n/a = not applicable. Blank boxes indicate no information available.

1 = Trophic linkages among or between species; 2 = Bioenergetics (balance between energy intake in the form of food and energy utilization by animals for life-sustaining processes) includes role of external factors such as global warming and pollution; 3 = Populations includes limiting factors and vulnerability to environmental changes; 4 = Ability to adapt/tolerate environmental changes.

Table 22. Matrix indicating state of information on the biology and ecology of slimy sculpin (*Cottus cognatus*) in areas south of 60°N.

Topics			Life Cycle Stages			
			Eggs	Larvae	Juveniles	Adults
Ecology and Habitat Requirements	Critical Habitat	Summer	n/a	L	L	H
		Winter	n/a			
	Reproduction	Timing	n/a	n/a	n/a	H
		Habitat	n/a	n/a	n/a	H
	Rearing and Refugia	Summer	n/a	L	L	M
		Winter	n/a			
	Food		n/a	L	L	M
	Water	Depth and Velocity		L	L	M
		Quality				L
Behavior	Feeding		n/a			L
	Predation and Predator Avoidance					M
	Migration		n/a			M
Physiology/Biochemistry	Growth		n/a			M
	Function[4]					L
Trophic Linkages[1] and Bioenergetics[2]						M
Populations[3]			n/a			L
Human Impacts						L

H = High; M = Medium; L = Low; n/a = not applicable. Blank boxes indicate no information available.
1 = Trophic linkages among or between species; 2 = Bioenergetics (balance between energy intake in the form of food and energy utilization by animals for life-sustaining processes) includes role of external factors such as global warming and pollution; 3 = Populations includes limiting factors and vulnerability to environmental changes; 4 = Ability to adapt/tolerate environmental changes.

WATERSHED ECOLOGY AND CUMULATIVE EFFECTS OF HUMAN INTERVENTIONS

In a critical review of "Ecological Development in Polar Regions" Dunbar (1968) suggested that consideration of three factors is critical for an understanding of Arctic ecology: the large seasonal oscillations in light and nutrient supply, the generally low productivity, and the immaturity of Arctic ecosystems. The Canadian Arctic is characterized by a short "growing" season (1-2 months), low summer temperatures (4-12 °C), reduced daylight in winter and perennial ice in many locations. (Woo and Winter 1993; Reist 1997). The abundant lakes and streams of the Canadian Arctic have low nutrient supplies and, as such, they have low productivity and biodiversity (Power 1997; Reist 1997; Reynolds 1997*a*, 1997*b*). The Arctic freshwater ecosystems are, therefore, vulnerable to environmental changes which may affect trophic relationships and nutrient regimes (Hammar 1989; Coulombe-Pontbriand et al. 1998).

In this section of the report a brief overview of the ecology and knowledge of Arctic freshwater ecosystems is presented with particular reference to the Slave Geological Province and the Coppermine River drainage. This section also provides a brief history of human interventions in the Coppermine River basin and the potential for their cumulative impacts on fish and fish habitat of the basin.

Hydrological considerations

The water cycle plays an important role in the climate, biology, and biogeochemistry of the Arctic and sub-Arctic regions (Vorosmarty et al. 2001). Hydrologic interactions with terrestrial and aquatic compartments and their biogeochemistry control all life in the pan-Arctic region. Most hydrological processes and related aquatic ecosystems (including the major Arctic rivers) in the Arctic and sub-Arctic are affected by snow and ice processes (Woo 1996). The hydrological regime of the Arctic is particularly susceptible to predicted climate change because of the dominance of the thermally-sensitive cryosphere (i.e. one of the Earth's spheres of irregular form existing in the zone of interaction of the atmosphere, hydrosphere and lithosphere, distinguished by the presence of water in the solid or super-cooled state) and its controlling influence on the water cycle. As such, a comprehensive understanding of the role of the hydrologic cycle in the Arctic and sub-Arctic is essential for assessing the impacts of changes (such as global warming), which may be brought about as a result of human interventions. Below we provide a brief description of the lacustrine and riverine habitats in areas north of 60°N.

Lake habitat: Most Arctic lakes are oligotrophic (i.e. fix less than 25 g $C/m^2/y$) , isothermal in the ice-free season, and experience a long period of ice cover each year. The duration of ice cover is dependent on lake size, exposure to wind and water exchange rate (Power 1997). Ice cover affects several processes in Arctic lakes. Of particular importance is the limited gas exchange with the atmosphere, especially in shallow lakes where oxygen depletion can lead to fish kills over winter (Barica and Mathias 1979). Meding and Jackson (2001) compiled data for 23 American temperate zone lakes to assess three alternative winter oxygen depletion models for estimating oxygen dynamics from freezing to thawing and found that: dissolved oxygen levels were constant or declined slightly for an average of 40 d after freezing and then declined rapidly; that once dissolved oxygen levels reached 3 mg/L, consumption slowed; photosynthesis and freeze-out estimates suggested that oxygen inputs are more important in shallow than in deep lakes; and dissolved oxygen decay rates (*k*) correlated with morphometry in shallow lakes, and with chlorophyll *a*, Secchi depth, and the sediment surface area to volume ratio in deep lakes. On the basis of these results, the authors (Meding and Jackson 2001) hypothesized that the failure of chlorophyll *a* to correlate with *k* in shallow lakes was because macrophytes were the primary source of decaying organic matter but they had not been included in assessments of winter dissolved oxygen depletion. As such, the authors concluded that some processes in deep lakes cannot be scaled down for small and shallow lakes.

It is also important to note that a considerable proportion of the annual heat budget of Arctic lakes is used to melt ice, which may also account for their low productivity (Schindler et al. 1974*a*, 1974*b*; Welch et al. 1987). For example, Welch et al. (1987) found that in Saqvaqjuac lakes (63°30'N), the annual heat budgets were slightly higher than for temperate lakes and increased with increasing mean depth. The annual surface irradiance penetrating the unfrozen water in these lakes was about 28% of total surface irradiance, which was about half that which would have been absorbed under ice-free conditions. The limnological research conducted by DFO in Saqvaqjuac area (63^o39'N, 90^o39'W, which is typical of the tundra landscape of the Canadian Arctic, has provided some knowledge of Arctic lake habitat (Welch 1985; Welch and Bergmann 1985*a*, 1985*b*; Bergmann and Welch 1985; Welch et al. 1987, 1988, 1989; Pike and Welch 1990; Bergmann and Welch 1990; Jorgenson et al. 1992; Hobson and Welch 1992; Hobson and Welch 1995). Some of the findings of this research are summarized below.

Most of these lakes are small (<50 ha), and those with a maximum depth of >2m do not freeze to the bottom (Welch 1985). The limnological characteristics of lakes in the Saqvaqjuac area are generally representative of the habitat characteristics of other lakes at similar latitudes in the Arctic (Welch and Bergmann 1985*a*). These lakes are "nominally cold monomictic, circulating once annually July-September, although temporary stratification frequently occurs in midsummer" (Welch 1985). They freeze around early October and become ice-free about July, or as late as August for the largest of the lakes (1,000-5,000 ha).

Dye experiments and measurements of conductance and temperature in small lakes of the Saqvaqjuac area showed that in winter "stored heat returning from the sediments warms adjacent water, which then sinks downslope. Water immediately beneath the ice moves laterally shoreward, picking up cryoconcentrated salts and sinking downslope (Welch and Bergmann 1985*a*). Displacement of deep waters upward at the lake centre was postulated by the authors as completing the circulation. In 2- to 10-ha lakes, the rates of water movement were in the order of 10 m/d, which the authors (Welch and Bergmann 1985*a*) believed could be expected to be ubiquitous throughout the Arctic. Results from the tritiated water and dye addition experiments as well as conductance and temperature profiles in these lakes showed that "during ice-on, the cold low density meltwater floated in a thin layer 0-100 cm beneath ice, extended over the entire subice-surface area, and left the lake without mixing with the heavier subice water" (Bergmann and Welch 1985*a*). On the basis of these results, the authors suggested that "lake models incorporating a lake flushing rate term need to be reevaluated to accommodate the lack of meltwater mixing beneath spring ice and more attention should be given to the early spring meltwater chemistry and its distribution within the upper lake strata."

The limnological characteristics of Arctic lakes show strong differences associated with latitudinal changes in biome characteristics, such as vegetation, permafrost, and regional climate (Rühland and Smol 1998). In a limnological study of 24 lakes located between Yellowknife (62°27'N, 114°21'W) and Contwoyto Lake (65°30'N, 110°00'W), Pienitz et al. (1997*a*, 1997*b*) reported that these lakes were mostly shallow (mean depth 8.2 m), poor in nutrients (3.4-12.7 μg total phosphorus/L; mean = 6.6 μg/L), with generally low alkalinity (pH 6.2-8.9) and low solutes (specific conductance near 0-100 μS/cm). Levels of all nutrients in lake water in this study showed a decreasing trend with increasing latitude (highest concentrations being in lakes with conifer-forested catchments in the southern part of the study area).

In a more recent limnological study of 56 lakes in the central Canadian Arctic, Rühland et al. (2003) reported similar latitudinal differences in limnology. The results of this study showed that boreal forest lakes generally have higher pH values (89% of lakes with pH values higher than 7.0) than Arctic tundra lakes where only 50% of pH values were higher than 7.0. Also, concentrations of major ions and other related variables, such as dissolved inorganic carbon, dissolved organic carbon, and nutrient concentrations particularly total nitrogen were higher in boreal forest lakes as compared to the Arctic tundra lakes.

River and stream habitats: Extreme cold in the Arctic creates different conditions for fish in streams and rivers compared with more temperate regions. These conditions include the formation of ice, limiting inputs of inorganic nutrients and carbon, and thermal effects on biological processes (Oswood 1997).

Icing of streams and rivers: Small and shallow streams freeze solid in winter and can only be utilized by fish in the short ice-free season. Thick ice cover also becomes a hindrance to the movement of fish. Ice formation in Arctic rivers plays an important role in the overall distribution and survival of fish. Frazil ice (formed as small crystals in riffle areas) can accumulate downstream and act as dams. These dams can lead to surface icing and downstream dewatering (Prowse and Gridley 1993, cited in Power 1997). According to Oswood (1997), "chemical conditions in the water beneath deep ice cover may be very inhospitable to biota, because of exclusion of dissolved material from the ice (increasing solute concentrations of the water) and because of low oxygen concentrations." Surface icing, depending upon its thickness, can cause severe depletion of oxygen as observed in many Arctic rivers (Schreier et al. 1980; Oswood 1997). Anchor ice (formed on stream beds in shallow riffles) can cause significant changes in stream flow (Power 1997), which in turn restrict movement of fish (Jakober et al. 1998).

Permafrost and watershed biogeochemistry: Although very little published information on watershed biogeochemistry is available from the permafrost areas of the Arctic, it has been shown that "permafrost has a profound and usually predictable influence on the hydrology of streams in the subarctic" (MacLean et al. 1999). The hydrologic regime of permafrost-dominated watersheds is characterized by low baseflows and high stormflows with a rapid onset following snowmelt (Woo 1986; Woo and Winter 1993). The infiltration of precipitation is much deeper in the absence of permafrost and, as such, it is thought to allow greater and more sustained baseflows and reduced stormflows (Woo and Winter 1993). In Alaskan taiga watersheds with extensive permafrost, MacLean et al. (1999) found that permafrost prevented deep percolation of water and, as such, they had higher fluxes of dissolved organic carbon, dissolved organic nitrogen and dissolved inorganic nitrogen. These authors also reported that the presence of permafrost reduces dissolved inorganic mineral loads in streams and, as such, its distribution influences hydrology, water temperature, and riparian vegetation (MacLean et al. 1999).

Smidt and Oswood (2002) showed that the presence or absence of permafrost in a watershed can play an important role in the distribution of stream macro-invertebrates. In Alaskan taiga forest streams, these authors found that mean macro-invertebrate abundance was significantly lower in stream reaches affected by the local presence of permafrost and highest in headwater streams unaffected by permafrost. Their results showed that "invertebrate abundance in similarly-sized headwater streams is strongly linked to the local presence or absence of permafrost and the taxonomic composition of macro-invertebrate communities is influenced by the quantity of watershed permafrost and stream size." They concluded therefore that permafrost is a major ecological factor "linking the macro-invertebrate community structure of stream reaches with landscape patterns at two scales: watersheds and reaches." Stream "reaches" have been defined as lengths of stream segments demarcated by geomorphic (e.g. channel slope) or landscape (e.g. riparian vegetation) discontinuities that contain a characteristic range of habitats and bed materials (Frissell et al. 1986).

Based on the important role played by permafrost in watershed biogeochemistry, it has been suggested that an understanding of interactions between permafrost and stream chemistry is important in predicting the effects of management, natural disturbance, and changing permafrost distribution on stream ecosystems and nutrient budgets in sub-Arctic watersheds (MacLean et al. 1999).

Jones et al. (2003*a*, 2003*b*, 2003*c*) studied spatiotemporal variation in the physical, chemical, and biological characteristics of pristine streams that represent a range of conditions near Lac de Gras in the Barrenlands region of the Northwest Territories. By using principal component analysis these authors were able to organize these streams into four groups on the basis of seven physical characteristics. While there were broad differences among these groups in physical characteristics, variation in chemical and biological variables was generally not large. But several biological and chemical variables were correlated with physical characteristics, in particular measures of stream size (such as bankfull width and depth, drainage area, and stream discharge). Annual variability in climate affected stream temperature and stream discharge and, as such, would have been expected to influence, for example, the growth of young of the year Arctic grayling. However, in an artificial stream, relatively lower temperature was not considered to be the primary factor responsible for the small size of Arctic grayling fry (Jones et al. 2003*a*).

The Coppermine River drainage: The Coppermine River flows in a northwesterly direction for about 845 km from its headwaters near Lac de Gras to its mouth in Coronation Gulf (Coulombe-Pontbriand et al. 1998; MacDonald 1999; MacDonald et al. 1999). As in other parts of the Arctic, the hydrology of this basin is influenced by a number of factors including regional climate, vegetation type, permafrost, geology and physiography (Coulombe-Pontbriand et al. 1998; MacDonald et al. 1999; MacDonald 1999).

The basin comprises two distinct geological regions, i.e. the Slave and Bear Geological Provinces. Although these two regions are part of the Precambrian Shield, they differ in their geological and physiographical features, which influence stream hydrology. According to Coulombe-Pontbriand et al. (1998), the southeastern and upper portion of the basin or the "lake district" is characterized by a chain of long, narrow lakes connected by short "turbulent" streams. These relatively large lakes play a significant role in regulating downstream flows (Coulombe-Pontbriand et al. 1998). These lakes (seven of them over 10,000 ha in surface area) store large volumes of water and provide a regulating effect on stream flows throughout the basin by delaying the annual peak flows or spring freshet, decreasing the magnitude of these peak flows, and by maintaining base flows throughout the winter (Wedel et al. 1988; Coulombe-Pontbriand et al. 1998). The headwaters of the Coppermine River are located in the vicinity of Lac de Gras, where according to Golder Associates Ltd. (1997), simulation studies suggest that the average lake level peaks in early November and is at its lowest in June. The rest of the basin (from the outlet of Rocknest Lake to the mouth) or the "river district" is primarily located within the Bear Geological Province and contains only one lake over 10,000 ha (Napaktulik Lake) and is dominated by the mainstem Coppermine River and its tributaries (Coulombe-Pontbriand et al. 1998).

The entire Coppermine River basin is located within the zone of continuous permafrost and is mostly above the treeline. Since continuous permafrost restricts seepage to a minimum, high volumes of surface water runoff are generated during snowmelt (Coulombe-Pontbriand et al. 1998). Surface runoff may amount to between 50 and 75% of total precipitation in some areas of the region (Acres Consulting Services Ltd. 1982, cited in Sly et al. 2001). Major differences in the seasonal flow of most rivers in the Coppermine River drainage have been reported (Acres Consulting Services Ltd. 1982; HBT-Agra Ltd. 1993, both cited in Sly et al. 2001). The causes of this seasonal variation include lowest precipitation during winter and ice formation at the bottom of some channels. In addition, evaporation over much of the region north of the tree line is in excess of 25% of the total precipitation. While lake storage plays an important role in reducing the variations in seasonal flows, inter-annual variability in flows is not uncommon (Acres Consulting Services Ltd. 1982; Wedel et al. 1988, both cited in Sly et al. 2001).

Although the Coppermine drainage lies in the continuous permafrost region of the Canadian Arctic, permafrost is usually absent beneath large lakes and rivers (Coulombe-Pontbriand et al. 1998). Soil moisture in permafrost exists in the form of ground ice. Permafrost can extend to depths of several hundred metres. In the Lac de Gras area, it extends to depths more than 270 m (BHP Diamonds Inc. and Dia Met International Ltd. 1995, cited in Sly et al. 2001) and near Contwoyto Lake (NWT) to depths of about 540 m (Tahera Corporation 2000, cited in Sly et al. 2001).

The two large lakes (Great Bear Lake and Great Slave Lake) play an important role in the hydrologic regime of the Coppermine River basin. These lakes, due to their massive volume, have an enormous capacity to attenuate the flow variability of the tributaries and stabilize outflows throughout the year (Coulombe-Pontbriand et al. 1998).

Aquatic biological considerations

The terrestrial, freshwater, and marine environments of the Arctic, as anywhere else, are intrinsically connected with each other in both space and time and, as such, they support a complex food web of plants, insects, fish, birds, and mammals (Sly et al. 2001). Because the total productivity of a region (ecozone) depends largely on the amount of energy that can be fixed by the primary producers (Hare and Thomas 1974), significant temporal and spatial differences in primary productivity have been observed in the Arctic. For example, the net annual production of terrestrial plant organic matter increases from 200 g/m^2 in tundra areas to 500 g/m^2 in taiga (forest communities found just south of the tundra), and 500 to 1,000 g/m^2 in boreal areas (Hare and Thomas 1974).

Net annual primary production in freshwaters of the Arctic is generally very low, largely because of the low nutrient concentrations (Ryder 1972; Brylinsky and Mann 1973). Rühland et al. (2003) assessed the chemical characteristics of 0.9-36.6 ha unnamed lakes (mean depth 3.5 m) influenced by climatic and climate-related factors such as degree of permafrost, vegetation, weathering, and duration of snow and ice. They determined that distinct differences existed among lakes in tree line ecozones, and that of all the measured variables, ionic and nutrient concentrations displayed the greatest changes from the sub-Arctic boreal forest to the Arctic tundra lakes: Arctic tundra lakes were more chemically dilute than sub-Arctic forested lakes. Because of low primary productivity in the fresh waters of the Arctic, the biodiversity is low, and in some lakes a trophic level may be represented by a single species (Power 1997; Johnson 2002). Thus, the trophic structure of the Arctic freshwaters is highly vulnerable to environmental change.

The generally simple, low biodiversity freshwater food webs in the Arctic are controlled by a complex interaction of both biotic (e.g. predation and competition) and abiotic factors (e.g. temperature, pH, DO, and landscape features such as lake and stream morphology) (Hershey et al. 1999; Jackson et al. 2001). The annual productivity is also low owing to a general lack of nutrients, especially in the tundra (Welch et al. 1989).

While solar energy input appears to have a greater influence on production than nutrient concentration on a global scale, nutrients generally assume greater importance in freshwater lakes within a narrow latitudinal range (Brylinsky and Mann 1973). In a limnological study of fertilized and natural lakes in the Saqvaqjuac area of the Canadian Arctic and their comparison with lakes in the Experimental Lakes Area (50°N, northwestern Ontario) and Char Lake (75°N, Canadian high Arctic), Welch et al. (1989) showed that "with increasing latitude, in small lakes, phytoplankton production decreases, phytoplankton production per unit light decreases less sharply, and the proportion of primary production occurring beneath ice cover increases."

During winter, Arctic lakes are sealed by unbroken ice cover and by permafrost surrounding the sides. Under these winter conditions all changes in free oxygen concentration are governed by internal processes (Welch and Bergmann 1985*b*). As such, photosynthetic activity is at its minimum or often non existent because of heavy snow cover and because of the latitude effect on day length. For example, in the

Saqvaqjuac area "the mean light flux penetrating the water in December ranges from 2 to 10 μE/m^2/d, depending upon snow cover, as compared with 900 μE/m^2/s in July" (Welch and Bergmann 1985*b*).

Primary production: Freshwater primary production in the Arctic is relatively low throughout the growing season (Power 1997). In deep Arctic lakes, such as Great Bear and Great Slave Lakes, the range of annual primary production is reported to be as low as 4 g C/m^2 (e.g. East Arm of Great Bear Lake) to about 40 g C/m^2 (e.g. Yellowknife Bay of the Great Slave Lake) (Fee et al. 1985; Johnson 1975*a*, 1975*b*, cited in Sly et al. 2001). Significant year to year variations in primary production have been reported in Arctic fresh waters, which appears to be related to the amount of nutrients in spring and summer runoff (Fee et al. 1985; Moore 1980).

In the large deep lakes (e.g. Great Slave and Great Bear Lakes), primary production is essentially phytoplankton based (Fee et al. 1985; Moore 1980). However, in clear and shallow lakes, where light can penetrate to the bottom, primary production is generally benthic due to the presence of benthic algae and periphyton (Moore 1979). In fact, in shallow lakes that have rooted aquatic plants, the contribution of these plants to primary productivity can be several times greater than that of phytoplankton (Johnson 1975*b*; Hecky and Hesslein 1995). The factors governing the distribution, abundance and productivity of macrophytes (Squires and Lesack 2003; Squires et al. 2002) and phytoplankton (Squires and Lesack 2002) have been studied in lakes of the Mackenzie River delta as part of a general program to assess their hydrology (Marsh and Hey 1989; Marsh and Lesack 1996). It was shown that macrophyte biomass increased with increasing water transparency and decreasing depth (Squires et al. 2002). However, in relatively turbid lakes, macrophyte biomass increased with the combined effect of increasing water color (decreasing water transparency) and increasing water depth. The authors suggested that "the increases in biomass with increasing water colour (coloured dissolved organic matter) and increasing depth, which together result in reduced light at the bed, may be explained by reduced exposure to ultra violet light." Squires and Lesack (2003) also studied the relationship between sediment nutrient content, macrophyte biomass, and community structure along a water transparency gradient among these lakes of the Mackenzie delta, and found that overall sediment organic matter (OM) and nitrogen content increased with increasing biomass of macrophytes, increasing OM and water clarity corresponded with increasing biomass of macrophytes in the lakes, and macrophyte community structure was shifted from dominance by erect *Potamogeton* at low and intermediate transparency and moderate sediment OM content to low-growing *Chara* and *Ceratophyllum* at high transparency and high sediment OM.

A recent study (Sierszen et al. 2003) of stable isotope food web analyses of two Arctic lakes (NE14 and I minus) in the Toolik Lake region of Alaska also suggested that the benthos was the primary source of carbon for adults of all species of both benthic and pelagic fish in these lakes. However, these authors "found no effect of turbidity, which may suppress benthic algae by shading, on food web structure" even though Secchi transparency varied between 10.2 m in Lake NE14 and 0.55-2.6 m in I minus Lake. These authors suggested "that the importance of benthos in the food webs of these lakes is due to their extreme oligotrophy, resulting in planktonic resources that are insufficient for the support of planktivorous consumers." They further asserted that their results "confirm the importance of benthic pathways, especially in oligotrophic systems where nutrient-mediated constraints on plankton may have implications beyond primary producers that extend to the structure and energetic foundation of whole food webs."

Bergmann and Welch (1990) have demonstrated that epilithic periphyton plays a moderate role in the nitrogen budget of small lakes in the Arctic. During the active nitrogen-fixation period (July to September 1981), periphyton in 0-2 m water depth was capable of fixing up to 0.86 mg $N/m^2/h$. This is a significant proportion of the total nitrogen budget, which increases with an increase in P loading and a decrease in the N:P ratio. Based on their results, these authors concluded "that Arctic periphytic N fixing communities are capable of adapting to changes in lake nutrient status."

Phytoplankton community Lac de Gras area: The studies (Moore 1978; BHP 1995, cited in Sly et al. 2001; Acres and Bryant Environmental Consultants Ltd. 1996, cited in Sly et al. 2001) conducted in the headwater lakes (Lac de Gras area) of the Coppermine River drainage suggest that the phytoplankton community in these lakes is dominated by chrysophytes (*Dinobryon cylindricum* and *D. bavaricum*), cyanophytes (*Lyngbya limnetica*), chlorophytes (*Ankistodesmus falcatus*), and bacillariophytes (*Cyclotella glomoreta*).

A study conducted in Lac de Gras reported a total of 40 algal species (Acres and Bryant Environmental Consultants Ltd. 1996). Studies conducted in the area have suggested that periphyton and macrophytes contribute significantly to the overall primary production of lakes in the Lac de Gras area of the Coppermine drainage (Kovats 1997).

The microbial loop and bacterial production: The microbial food webs composed of bacteria, autotrophic picoplankton, protozoans, and microzooplankton play an important role in the overall trophic dynamics of lakes (del Giorgio and Cole 1998). It has been shown that pelagic bacteria are numerous and compete with phytoplankton for nutrients and use algal exudates as a source of carbon. Thus, bacteria in freshwater ecosystems are not mere mineralizers of detritus but also represent an important food chain component or "microbial loop" (Crump et al. 2003).

Several studies have shown that the microbial loop may be responsible for more of the carbon and energy production and transfer in the pelagic zone of many lakes than the "classical" phytoplankton-mesozooplankton food chains (Stockner and Shortreed 1989; Berman 1990; Weisse et al. 1990; Weisse and MacIsaac 2000; Crump et al. 2003). There is also evidence that in oligotrophic lakes, bacterioplankton and microbial food webs are responsible for a larger share of the overall production and energy processes of the pelagic community than that which occurs in more productive lakes (Cole et al. 1988; Simon et al. 1992; Cole and Caraco 1993; Weisse and MacIsaac 2000).

In their study of the bacterial production in highly oligotrophic British Columbia lakes, Weisse and MacIsaac (2000) reported that bacteria contributed about 24% to the phytoplankton-bacteria carbon biomass, but their relative contribution declined to <11% with increasing lake productivity. In these lakes, the abundance of heterotrophic nanoflagellates (HNF) was positively correlated with bacterial numbers. In grazing experiments, it was shown that HNFs were the primary pathway for transferring bacterial production to higher trophic levels, and predation on HNFs by ciliates and rotifers (microzooplankton community) appeared to exert "top-down" control over the abundance of HNFs and the transfer of carbon from bacteria (Rublee 1998). The HNFs were, in turn affected by the abundance of the mesozooplankton community (principally copepods). Similarly, in a highly oligotrophic tundra lake (Toolik Lake, Alaska) with low productivity (around 15 g $C/m^2/y$), Hobbie et al. (1998, 1999) demonstrated that HNF grazing removed 14% of the bacteria per hour (an amount roughly equivalent to bacterial production), indicating "top-down" control of bacteria. The authors concluded that "this limitation of top-down control is caused by the oligotrophic conditions that allow only small numbers of zooplankton and microzooplankton to exist." These investigations suggest that highly oligotrophic (i.e. ultra-oligotrophic) Arctic lakes "eliminate the superimposed variability of food web structure and provide an example of the importance of resource availability at the extreme end of nutrient and temperature gradients" (Hobbie et al. 1998, 1999; Rublee 1998).

Secondary production: Mechanisms affecting zooplankton community dynamics are particularly subtle and complex in harsh environments such as the Arctic (Kettle and O'Brien 1978; O'Brien et al. 1979). The commonly-accepted theories of zooplankton species distribution hold that (a) large-bodied zooplankton species are excluded by fish predation and are found only in lakes and ponds without fish, and (b) because small-bodied species are unable to compete successfully against large ones and are also preyed upon heavily by invertebrate predators, they exist primarily in lakes with fish. However, O'Brien et al. (1979) reported that this pattern is not found in a group of lakes and ponds in Arctic Alaska. In some of these Alaskan lakes, both large and small zooplankton species coexisted with populations of facultative planktivorous fish.

In a more recent and comprehensive study of physical, chemical, and biotic effects on Arctic zooplankton communities and diversity involving 104 Alaskan Arctic lakes, O'Brien et al. (2004) have shed more light on the complexity of interactions between fishes and zooplankton in Arctic lakes. The major findings of this study were that "Lake depth was related to the distribution of zooplankton in that smaller zooplankton species were found in deep lakes and large species in shallow lakes; neither ionic strength nor chlorophyll was a major factor in zooplankton presence or absence; when fish were present, few, if any, very large zooplankton occurred; the relationship between the presence and density of smaller-sized zooplankton species and the presence of specific species of fish was complex and did not fit the anticipated patterns; the diversity of zooplankton in these Arctic lakes was not related to chemistry or fish distribution, but species richness increased with increasing lake area and depth, and for a few zooplankters (notably *Holopedium gibberum*), the fraction of lakes containing the zooplankter varied by region."

Although the results of this study showed a low effect of fish species on zooplankton distribution, the effect of Arctic charr, and in particular slimy sculpin, on zooplankton distribution was completely unexpected as these fish species are not considered to be highly planktivorous. A statistical analysis that examined whether the presence of each zooplankter species was dependent on the fish communities, revealed that lakes with only charr and sculpin were much less likely to have the zooplankter *Heterocope septentrionalis* and more likely to have *Daphnia longiremis* than any other lakes. The authors, therefore, recommended that "even with the relatively low diversity in these

Arctic lakes, more research is needed to clarify the interactions between fishes and zooplankton."

The complexity of trophic relationships in lake ecosystems has been shown by Persson et al. (2003). They revealed the occurrence of a dynamic trophic cascade in which the community, over time, alternated between two different configurations: the size structure of the fish (*Perca fluviatalis*) changed from a dominance of stunted individuals to a dominance of gigantic cannibals among adult individuals. The authors showed that these intrinsic size-structured changes in perch populations were the driving force behind distinct abundance switches observed in zooplankton and phytoplankton. The authors concluded that the "vital ingredients involved in the observed population dynamics are variable intensities in intercohort cannibalism (older perch *vs* YOY perch) and intercohort competition (1-year-old *vs* YOY perch)."

The role of fish communities in regulation of the structure and function of freshwater ecosystems (lakes and streams) was earlier reviewed by Northcote (1988) who acknowledged the complexity of trophic dynamics in freshwater ecosystems and remarked that "it may be naïve and simplistic to expect the structure and function of freshwater ecosystems to be "controlled mainly by "top-down" or "bottom-up" processes." Thus, he asserted that "both these processes can have many important effects on each other and both must be involved in (trophic structuring of) freshwater ecosystems" in varying degrees.

A limited number of studies (Moore 1978; Acres and Bryant Environmental Consultants Ltd. 1996) conducted in the upper Coppermine River drainage suggest that biodiversity of zooplankters is generally low. Overall, the zooplankton community in these headwater lakes appeared to be limited to three groups of taxa, namely rotifers (*Kerateela cochlearis, Kellicotia longispina, Conochilus hippocrepic*), copepods (*Cyclops scutefer, C. bicuspidatus, Diaptomus sicilis*), and cladocerans (primarily *Holopedium gibberum*). Surveys in Lac des Gras suggested that 80% of the zooplankton community was represented by rotifer species of the genus *Conochilus* (Acres and Bryant Environmental Consultants Ltd. 1996)

Fish communities: Biodiversity of freshwater fish is low in the Arctic and according to Power (1997) they represent less than 1% of the known species of fish in the world. According to Reist (1997), only 42 freshwater fish species are represented in the Canadian Arctic, 16 of which are salmonids and half of these salmonids demonstrate some degree of anadromy. McPhail and Lindsey (1970) have, however, reported 60 freshwater and anadromous fish species from northwestern Canada and Alaska.

The fish biomass is generally low in Arctic freshwater systems, and in undisturbed and unexploited lakes, the fish biomass is usually restricted to small numbers of larger and older fish, representing the conversion and storage of many years of basic production (Ryder 1972; Johnson 1973; DeGraaf and Machniak 1977; DeGraaf 1988; Sly et al. 2001). In general, lake trout and lake whitefish are the dominant species in Arctic lakes and account for nearly 95% of the fish biomass (Johnson 1976). The dominance of large, old fish in catches from these Arctic lakes has, however, been debated in the literature and its cause has been variously hypothesized. According to Johnson (1976), large fish in Arctic lakes dominate and suppress the younger and smaller individuals and, as such, the younger and smaller fish grow slowly, and enter the dominant class only when space becomes available due to natural mortality in the dominant class. Upon entering the dominant class, they are reported to assume a rapid growth pattern (Johnson 1976). Power (1978), however, in his evaluation of unexploited northern fish populations, rejected Johnson's explanation for the predominance of older and larger individuals in Arctic lakes. He proposed that the growth in these populations is relatively rapid until maturity, when it slows down through the life span which often exceeds 50 years. In addition, mortality declines rapidly through early life and stabilizes to a very low level through much of the life span. These growth and mortality patterns produce many small, few intermediate, and many large fish in a population. Power (1978) attributed the unimodal samples of large fish or a bimodal sample of large and small fish from such populations to selection by gill nets.

The Coppermine River drainage has the highest numbers (at least 18) of fish species compared to other systems in the region (MacDonald et al. 1999). Lake trout are the most commonly-occurring and abundant fish species in the Coppermine River Basin. Arctic grayling, round whitefish, longnose suckers and northern pike are also broadly distributed. Other fish species of importance are lake whitefish, broad whitefish, cisco, least cisco, burbot, slimy sculpin, lake chub, Arctic charr, ninespine stickleback and inconnu (MacDonald et al. 1999). It has been generally observed that individual lakes appear to be dominated by just a few fish species.

The structure of fish communities varies considerably among Arctic lakes (Bodaly et al. 1989). Oswood et al. (2000) examined the spatial distributions of Alaskan freshwater fishes using two ecoregions (defined by climate, vegetation, and soils) and hydroregions (or catchments). They found that "similarities of fish faunas were highest in adjacent hydroregions, with declining similarity between latitudinally disjunct hydroregions." As such, these authors hypothesized "that the distribution of Alaskan freshwater fishes is complexly determined by eco-physiological requirements of fishes along the latitudinal gradient from northern rain forest to Arctic tundra, by current and past barriers to dispersal, and by legacy of Pleistocene glaciation."

Fish-food web interactions: In the freshwater food webs of the Arctic, a number of small fish species (e.g. troutperch, lake chub and ninespine stickleback) provide a key link between upper and lower parts of the food web (Scott and Crossman 1973; BHP Diamonds Inc. and Dia Met International Ltd. 1995, cited in Sly et al. 2001). Hershey et al. (1999) have explained the fish community structure and trophic dynamics in Arctic lakes based on a "geomorphic-trophic hypothesis," which proposes a unifying conceptual model of the Arctic landscape. This model "integrates lake processes into their geomorphic setting by illustrating how the stream network constrains or channels the dispersal of fish and links fish dispersal with fish control of lake food webs" (Hershey et al. 1999). Thus, the authors argue that the landscape, in which these lakes are embedded, indirectly controls trophic structure of lakes "because landscape controls the distribution of fishes, and fish control lake trophic structure."

Based on the template provided by the "geomorphic-trophic hypothesis" for Arctic lake food webs, Hershey et al. (1999) have identified six different types of fish communities in Arctic lakes in the vicinity of Toolik Lake (North Slope of Alaska) as follows: (a) lakes dominated by lake trout (*Salvelinus namaycush*), but also usually containing grayling (*Thymallus arcticus*), slimy sculpin (*Cottus cognatus*), burbot (*Lota lota*), round whitefish (*Prosopium cylindraceum*), and rarely Arctic charr (*Salvelinus alpinus*); (b) lakes containing Arctic charr and slimy sculpin; (c) lakes with Arctic grayling and slimy sculpin; (d) lakes with Arctic grayling only throughout the year; (e) lakes with grayling in the summer only; and (f) lakes with no fish.

According to Hershey et al. (1999), the absence of fish in a lake happens because the stream outflow gradient is inaccessible to fish or the lake is of insufficient depth to support fish and is not connected to a deeper lake. Similar differences in fish community structure were demonstrated in the vicinity of BHP Billiton's Ekati diamond mine, where several lakes were dewatered and the entire fish fauna of each lake was identified and complete population counts were obtained for each species (BHP 1997). The results showed that the structure of fish communities varied considerably from lake to lake. For example, in Panda Lake, lake trout and round whitefish were dominant and their population size was similar, but in nearby Airstrip Lake, burbot and lake trout were the dominant species. In the third dewatered lake, Misery Lake, lake trout was the only fish species present.

On the basis of the "geomorphic-trophic hypothesis," Hershey et al. (1999) have also attempted to explain the food web interactions in Arctic lakes. To test this idea, these authors surveyed 110 lakes in Alaska over the summers of 1997-1999, and on the basis of the presence and absence of fish and zooplankton species, they concluded that lakes which lacked fish almost always had large-bodied zooplankton. Some lakes with fish, however, also had these species; lakes with fish had small-bodied zooplankton, however, some lakes lacking fish also had these species; one species, *Cyclops scutifer* which occurred in 94% of the lakes was impacted little by landscape setting; the fish species also had important effects, for example, *Heterocope septentrionalis* and *Daphnia middendorffiana* were more likely present in lakes that contained lake trout, but these species were unlikely in the presence of Arctic grayling; *Heterocope* was also less likely to be present in lakes with Arctic charr while *Daphnia longiremis* was more likely to be present.

Hershey et al. (1999) further concluded that the presence of fish can alter the biomass and composition of zooplankton communities, which in turn can affect energy flow from phytoplankton to zooplankton. For example, these authors found that the mean zooplankton biomass in a fish-less lake was 4 times higher than the biomass in lakes with fish.

It is important to note that in recent years considerable empirical evidence has been provided to support the role of fish as integrators of benthic and pelagic food webs in lakes (Vander Zanden and Vadeboncoeur 2002; Sierszen et al. 2003). Vander Zanden and Vadeboncoeur (2002) synthesized data on the diet from 470 fish populations (15 species) and stable isotope data from 90 fish populations (11 species) of north temperate lakes. They showed that benthic secondary production plays a central role in supporting higher trophic level production (i.e. across all species considered, zoobenthos averaged 50% of total prey consumption). These authors, therefore, concluded that "recognition of the duality of pelagic and benthic production pathways, which are closely linked by mobile consumers such as fishes, will provide an enhanced energetic template for understanding lake trophic dynamics." Sierszen et al. (2003) similarly provided evidence for benthos as the basis for Arctic lake food webs. Stable isotope food web analyses of two Arctic lakes (10-15 m deep) in the Toolik Lake region of Alaska by these authors provided strong indications that "benthos are the primary source of carbon for adults of all species of benthic and pelagic fish species present." Thus, it has been suggested that there is a need to take "a more integrated, whole-ecosystem perspective of lakes that considers the quantification of energetic flows as a precursor to an understanding of food web dynamics and lake ecosystem functioning" (Vander Zanden and Vadeboncoeur 2002). Stable isotopes of sulphur, carbon, and nitrogen have also been shown as good indicators of trophic level and fish migration in the lower Mackenzie River Basin (Hesslein et al. 1991).

Fish strategies for survival in the Arctic environment: For Arctic and north temperate freshwater fish, winter is probably the most critical and limiting factor for survival, and many Arctic fish respond to this challenge through adaptative behaviors (Reynolds 1997*a*; Thorpe 1994; Power 1997). These include seasonal migrations associated with metabolic requirements for feeding, growth and survival, spawning, and avoidance of extreme winter conditions (Power 1997; Reist 1997; Northcote and Hinch 2004). McDonald et al. (1992) commented that the occupation of nearshore, less food-rich refuge habitats by young fish has implications to their survival such that they may "pay a substantial penalty" (predation in winter and starvation) if food is scarce (the productivity of the littoral zones of Arctic lakes differing from those of more temperate regions which do not experience extensive ice cover for many months of the year).

Annual migrations of fish to summer feeding grounds have been reported for many Arctic-resident fish (Reynolds 1997*a*) and considerable fidelity to these feeding grounds has been demonstrated for Arctic grayling (Northcote 1995; Buzby and Deegan 2000). With the advent of winter and declining water temperatures, the fish move to over-wintering refugia (thereby avoiding anchor ice and total freeze up in the case of smaller and shallower streams), where they adopt different behaviors and respond physiologically to environmental changes. For example, the territorial behavior that is so common during summer feeding is replaced by aggregation behavior (Reynolds 1997*a*). Acclimation to winter conditions means high metabolic costs to fish. Following an initial acclimation period at the onset of winter, however, metabolism is relatively low and more stable which enables the fish to condition themselves to slow habitat deterioration over the rest of the winter months (Cunjak and Power 1987). Generally, the physiological response of fish to winter conditions includes decline in serum proteins and body lipid (Cunjak and Power 1986), decline in energy reserves (Dutil 1986), and reduced protein synthesis (Bulow et al. 1981).

In many areas of the Canadian Arctic, rivers and streams are covered with ice and develop ice on the streambed for significant periods of the year. These conditions reduce the habitat suitability of certain portions of these systems for over-wintering juvenile and resident fish (Cunjak 1996). In some areas, groundwater upwelling in side channels and sloughs, however, provides over-wintering refugia. Similarly, coldwater fish such as lake trout in more southerly temperate regions (e.g. the Great Lakes watersheds) seek cooler groundwater upwelling in summer when relatively shallow lakes become isothermal (Snucins and Gunn 1995). These groundwater upwelling refugia, therefore, should be condidered for the survival of fish populations which depend on them (Alfredsen and Tesaker 2002; Morse and Hicks 2005).

There is a general paucity of information about the over-wintering habitat of fishes in sub-Arctic and Arctic lakes. These lakes are ice-bound for long periods, often more than half the year, and present very different and difficult environmental conditions in the winter compared with the open-water period (Klemetsen et al. 2003). Depending on the latitude, more food is available during a brief (1- to 5-month) period (spring-fall) and little or no food is available during the winter months (Reynolds 1997*a*). As such, strategies are required by fish to conserve energy during winter and to ensure the adequacy of energy reserves for over-winter survival. For example, Arctic charr (*Salvelinus alpinus*) populations in two Finnish lakes (69°N) "perform regular habitat shifts between the littoral zone in the winter, the profundal zone at ice break, and the whole lake in the summer and autumn" (Klemetsen et al. 2003). These lacustrine Arctic charr populations concentrated in the littoral zone (0-15 m) of lakes during the entire winter (December to May) despite very low temperatures ranging between 0.2 and 0.7 °C. The possible reasons for this behavior given by the authors included high prey availability, low predation and competition, and comparatively better light under snow and ice in shallow compared with deep water (Schindler et al. 1974*a*). But further north (north of the Arctic Circle i.e. 66°33'N) there is some evidence which implies that Arctic charr do not feed most of the winter (Welch 1976; Parker and Johnson 1991), that many live at the "edge of existence," and are able to withstand up to 2 years of fasting in this frigid environment (Johnson 1975*a*, 1975*b*, 1975*c*).

Winter causes a variety of physiological changes in fish (Cunjak 1988). At the higher latitudes fish are subject to seasonally-shorter periods of growth potential followed by a long over-wintering period with greatly-limited availability of food resources. Thus the ability of fish, particularly juveniles, to acquire sufficient growth and energy reserves to survive winter can be constrained (Cunjak 1988; Post and Parkinson 2001). Among underyearlings, smaller fish tend to be more susceptible to over-winter mortality due to relatively higher basal metabolism and consequential increased depletion of energy reserves (Shuter and Post 1990; Post and Parkinson 2001). A major source of winter mortality in underyearling fish is depletion of lipid reserves (Pratt and Fox 2002; Biro et al. 2004). Over winter, declines in serum protein levels, body lipid content (Cunjak and Power 1986), energy reserves (Dutil 1986), and depression of tissue RNA concentrations, an indicator of protein synthesis (Bulow et al. 1981), have been reported. Metabolic costs are high in early winter (during initial acclimation) but later metabolism is lower and more stable (Cunjak et al. 1986; Cunjak and Power 1987; Cunjak 1988), unless required to counter stressful conditions (hypoxia, metals) that elevate metabolism causing a more rapid depletion of energy reserves and mortality (winter stress syndrome, Lemly 1993, 1996).

Reynolds (1997*a*) points out that because of the variations in the severity of winter in the Arctic (Alaska), fish have adopted two general survival strategies, that is, spring spawning and summer to fall spawning. According to Reynolds (1997*a*), "spring spawners" devote energy to gamete production during the previous summer or winter, but invest less energy in doing so; their eggs are smaller and hatch in early summer. Spring spawners are "free" to spend most of the summer feeding to restore and perhaps add to body reserves lost during winter. However, their young must grow enough during summer to survive their first winter. "Summer to fall spawners" devote more energy to egg production, generally produce larger eggs that over winter and hatch in the substrate, with the young living on yolk before they emerge. According to Reynolds (1997*a*), despite feeding advantages in spring and early summer, which the "Summer to fall spawners" pass on to their young as energy for winter survival, adults must face winter in reduced body condition. The adult response, therefore, is to spawn once and die (e.g. Pacific salmon), spawn once every 2 to 3 years (e.g. charr), or produce large quantities of small eggs (e.g. round whitefish) (Reynolds 1997*a*).

In a comprehensive review of the winter habitat of selected stream fishes and potential impacts from land-use activity, Cunjak (1996) concluded that "Stream fishes have adopted many adaptive mechanisms to meet the demands of winter. The complexity and dynamic nature of a stream in winter must be viewed from a spatial, as well as temporal, aspect. That is, winter habitat suitability can, depending on individual preferences and local environmental conditions, be achieved in any number of habitat types, stream reaches, and basins. Habitat suitability and use can vary with time over the course of winter (e.g. early winter vs. late winter vs. midwinter; day vs. night). Such complexity suggests fragility in the inter-relations between animal and habitat in winter. The frequency and prevalence of significant winter mortalities seem to emphasize this fragility. Assurance of winter habitat protection for the many life stages and species that comprise a fish community will not be an easy task for fisheries managers. Because of the variety of winter habitat preferences and the long distances that some fishes travel between summer and winter sites, the most obvious advice is to maintain habitat complexity. In the case of degraded streams, winter habitat enhancement needs to restore complexity. That each stream is different and unique is known to any field biologist, but an unwelcome statement to a manager trying to develop habitat guidelines with broad geographic applicability. However, there are some generic rules regarding winter habitat which apply to all streams. These rules include the understanding that diverse, complex microhabitats and the corridors that allow access between them are essential for over-wintering fishes; summer and winter habitat requirements may be different; groundwater discharge zones provide winter refugia; ice can markedly influence habitat suitability; and land-use activities that increase sediment loading and reduce stream flow can deleteriously alter availability of winter habitats. Quantitative data demonstrating the importance of winter habitats, and contrasting survival under different winter conditions, are largely lacking and can only be obtained by applying the type of effort previously reserved for the spring-autumn period. Not until we consider winter as part of the field season rather than a time to work up summer data, will we be able to effectively conserve and manage fish habitat."

It is clear from the literature reviewed that winter is a critical and limiting period for fish production in both Arctic and temperate regions. Thus, we concur with the suggestion that winter habitat assessment procedures for fish need to "include a study of fish behavior in winter conditions, the effects of various ice types on habitat selection, and the dynamics of ice breakup on the microscale" (Alfredsen and Tesaker 2002). In northern Canada, rivers and streams are covered with ice and develop ice on the streambed for significant periods of the year. These conditions reduce the habitat suitability of certain portions of these systems for over-wintering juvenile and resident fish (Cunjak 1996). In some areas, however, groundwater upwelling in side channels and sloughs provide over-wintering refugia for fish. Similarly, coldwater fish such as lake trout in more southerly temperate regions (e.g. the Great Lakes watersheds) seek cooler groundwater upwelling in summer when relatively-shallow lakes become isothermal (Snucins and Gunn 1995).

Protection of these refugia are, therefore, important for the survival of fish populations which depend on them (Alfredsen and Tesaker 2002; Morse and Hicks 2005).

Limnological studies of lakes in the Lac de Gras area

The limnological and biological characteristics of two relatively-undisturbed lakes in the Lac de Gras area, NWT, ("Fish 1" (maximum depth 6 m, surface area 45.2 ha) and "Fish 2" (maximum depth 18 m, surface area 68.0 ha)) were studied by Martin (2001). In these lakes, "ice-off" occurred around the third week of June, and the lakes temporarily stratified during the open-water period in three sampling years. Summer concentrations of total phosphorous ranged from 4-16 µg/L and total N ranged from 99-396 µg/L. Chlorophyll *a* averaged around 1.3 µg/L for most of the sampling period for both lakes and all depths. The only exception was following ice out in Fish 2 Lake in 1996 when concentrations of chlorophyll *a* averaged 6.7 µg/L at 1 m and 15.1 µg/L at 4 m water depth. These levels were not matched in Fish 1 Lake in 1996, and the period was not sampled during 1995 (Martin 2001).

Zooplankton communities were sampled in both lakes, and cyclopoid nauplii were the most important components of the communities, based on numbers of individuals/L over both years, followed by calanoid copepodites. Chironomid emergence began immediately after ice out, reached peak levels shortly thereafter, and continued throughout the summer season at a lower level. Both lakes have populations of lake trout, whitefish, Arctic grayling, and burbot. Cyprinids and sculpins were present in the lakes, but there was no attempt to sample them. Little information was collected for burbot. Round whitefish dominated the fish community in both lakes. Fish 2 Lake supported a larger population of lake trout than Arctic grayling, which was the opposite of the community structure in Fish 1 Lake. Both round whitefish and Arctic grayling populations were dominated by smaller size classes. Larger individuals dominated the lake trout populations. The diets differed between species, although zooplankton was the major diet item consumed by all species. Lake trout also fed on chironomids and fish. Other prey items included several species of aquatic insects, molluscs, notostracids, and terrestrial insects. Arctic grayling also consumed chironomids, and to a lesser extent several aquatic insects, terrestrial insects, and notostracids. Round whitefish relied less on zooplankton than the other two species, and consumed proportionally more chironomids, notostracids, trichopterans, and molluscs. Within a given lake, (and in relation to the species captured) Arctic grayling were positioned at the lowest trophic level, and lake trout at the highest. Round whitefish were intermediate between the two, although they tended to be less depleted in ^{13}C, likely as a result of the greater importance of benthic invertebrates and the lower reliance on zooplankton for this species compared to the other two. Streams allowed access between the two lakes and were used by Arctic grayling, lake trout, burbot, and sculpin.

The results of the study by Martin (2001) also showed high levels of Cu in liver tissue of fish, suggesting exposure to this metal. Martin (2001) also reported that levels of Se in both muscle and liver tissue were close to the toxic effect threshold for overall health and reproductive vigor in fish, and Hg was above the guidelines for subsistence fish consumption in the larger lake trout sampled. All other metals analyzed were within the range found in other studies in the Northwest Territories.

Based on his results, Martin (2001) recommended continued monitoring of the contaminant levels of fish from lakes in the area to ascertain changes resulting from the redistribution of elements as a consequence of ongoing mining operations. He also recommended further work to determine the importance of stream habitats to fish populations because of their importance for the movement of fish between lakes.

Socio-economic considerations

The use of renewable biological resources is fundamental to the economic and socio-cultural well being of many northern societies and cultures (Schlosser et al. 2003). The Northwest Territories and Nunavut are largely undeveloped and the Aboriginal people of the region are still carrying out their traditional life style of hunting, trapping, and fishing for subsistence (Sly et al. 2001; MacDonald et al. 1999). Indigenous people have been living in this area and have utilized its resources for thousands of years. Watersheds of the region support a variety of fish which provide food for these people (MacDonald et al. 1999).

The Canadian north (NWT and Nunavut) has a long history of mining activity which dates back to the late 1800s and early 1900s when precious and semi-precious metals were mined at several locations (Lemly 1994). About 30 ongoing mineral exploration projects were reported in 2001 (Sly et al. 2001). Uranium was extensively mined in the Northwest Territories from 1939 and some mines were in production for 30 to 40 years (Lemly 1994). Abandoned mines and improperly-stored tailings are the legacy of the past mining practices and continue to be of concern because of contaminants that continue to leach from the tailings (Lemly 1994). It has been estimated that there are approximately 37 abandoned mines in the Northwest Territories alone, and as many as 120 abandoned mines in the Yukon Territory (MacKasey 2000). Acid mine drainage from tailings is a continuing environmental problem. It is of potential concern that both historical and more recently-deposited mine tailings contain high levels of arsenic (up to 25,000 ppm in the Conn Mine tailings, Yellowknife, NWT (MiningWatch Canada 2001)).

The potential for further development in the region is high; it has witnessed heightened mineral development activity in recent decades (Sly et al. 2001). With the discovery of diamonds in the early 1990s, large areas of the region have been staked for mineral exploration (Stiff 2001).

Diamond mining activity is largely concentrated in the Slave Geological Province (SGP), a tract of 19,500,000 ha, which straddles both the Northwest Territories (NWT) and Nunavut. Since the discovery of diamonds at Point Lake (NWT) in 1991, the diamond mining sector in Canada has been steadily growing (Sly et al. 2001). The Ekati Mine has employed over 700 full time workers and purchased around $356,000,000 worth of goods and services each year (Indian and Northern Affairs Canada 2001). Similarly, the Diavik diamond mine which started production in 2003 had employed over 1,000 people to work on the project. It is anticipated that following 2006 and 2007, when both Tahera Corporation's Jericho project and De Beers Canada Mining Inc.'s Snap Lake project are respectively expected to be in production, Canada will produce between 12% (Sly et al. 2001) and 15% (Santarossa 2004) of the world's supply. Thus, diamond mining in northern Canada appears to have far-reaching implications for the regional economy and the environment.

Cumulative watershed level effects

Cumulative effects are essentially the changes in an environment caused by the combination of past, present and reasonably-foreseeable future actions. It is generally agreed that almost all impacts are influenced by multiple activities and, as such should be evaluated in the context of cumulative impacts rather than as individual impacts (Reid 1998; Schindler 1998). While this holistic approach is a logical goal for those charged with the responsible management of large ecosystems, the complexity of the changes, the knowledge of that which is impacted and the extent of change create a challenge that is formidable in terms of its likely outcome. The increasing complexity of systems from simpler components to the more complex create increasing risk and uncertainty with respect to the prediction and/or determination of outcomes (refer to section "Information and ecosystem complexity" and Figure 24).

A widely-accepted legal definition of cumulative impacts is the one given by the United States Council of Environmental Quality (CEQ Guidelines, 40 CFR 1508.7 issued 23 April 1971). "Cumulative impact" is the impact on the environment which results from the incremental impact of the action when added to other past, present, and reasonably-foreseeable future actions regardless of what agency (Federal or non-Federal) or person undertakes such other actions. Cumulative impacts can result from individually-minor but collectively-significant actions taking place over a period of time.

Cumulative effects are the combined effects of multiple activities in both time and space, and "cumulative watershed effects" are those effects which involve processes of water transport, such as effects on hydrology, sedimentation, organic material and chemicals (Reid 2001). In this context a "watershed" and an "ecosystem" can be considered equivalent, but, any watershed may encompass multiple "ecosystems" (such as a series of lakes), and a sufficiently-large ecosystem can encompass multiple watersheds (such as river systems draining into large lakes such as any of the Laurentian Great Lakes).

Many regions of Canada's north including the Coppermine River basin have been confronted with increasing development pressure in recent decades (Lemly 1994; MacDonald et al. 1999; AXYS Environmental Consulting Ltd. 2000; MacKasey 2000; Sly et al. 2001). As such, concerns have been expressed with respect to the potential of cumulative effects of human developments in both the temporal and spatial contexts. According to the Cumulative Impact Monitoring Program (CIMP) (2005) a number of human interventions are affecting the NWT environment at the present time, including infrastructure development, mineral exploration and development (including base metals and diamonds in the eastern NWT), oil and gas exploration and development (western areas of NWT), electrical generation, tourism, exploitation of renewable natural resources (e.g. fishing, hunting), contaminated sites (abandoned mines, waste sites), long-range transport of atmospheric pollutants, and climate change. The scope of these interventions has been briefly provided (within the Socio-economic considerations section, above) and further details can be found in MacDonald et al. (1999) and Sly et al. (2001). Of major and immediate concern, however, are resource exploration activities relating to diamond mining in the Slave Geological Province, oil and gas activity in the Deh Cho and

Inuvialuit Regions, and the major pipeline corridor along the Mackenzie Valley (CIMP 2005). According to this report these development activities are likely to contribute to cumulative impacts in the region which would necessitate an adequate understanding of the baseline environmental conditions against which to predict and assess change.

Implications of global environmental changes: Assessment of environmental stressors associated with development needs to be done with full consideration of the cumulative effects, global environmental changes such as global warming, increasing UV levels associated with ozone thinning, and long-range atmospheric transportation of persistent organic pollutants (Schindler et al. 1990; Schindler 2001; McCarty 2001; Häder et al. 2003).

Global warming: Scientific evidence is mounting with respect to global warming as a result of a continuing increase in levels of carbon dioxide and other "greenhouse gases" in the atmosphere (Kerr 1996; Nelson 2003; Arctic Climate Impact Assessment, ACIA 2004). Climate models have predicted significant increases in air temperature and decreases in soil moisture over several decades in North America as a result of increasing accumulations of greenhouse gases in the atmosphere. The earth's climate has already warmed by 0.5 °C over the past century, and recent studies have shown that it is now possible to detect the effects of a changing climate on ecological systems (McCarty 2001). Some models have predicted that, in North America, the greatest effects in summer will occur at about 48°N to 52°N, where summer temperature increases of up to 9 °C and >50% decrease in soil moisture may occur over the next few decades (Schindler et al. 1990).

In Canada, 20 years of climatic, hydrologic, and ecological data in the experimental lakes of northwestern Ontario have demonstrated a 2 °C increase in air and lake temperatures, a three-week increase in the ice-free season, and higher than normal evaporation and lower than average precipitation, which have decreased rates of water renewal in lakes (Schindler et al. 1990). The results of these long-term studies have also shown an increase in populations and diversity of phytoplankton but without any consistent trends in primary production. Deepening of lake thermoclines has reduced the summer habitat for organisms such as lake trout and opossum shrimp (Schindler et al. 1990).

Similarly, in Alaska, based on a 16-year record of physical, chemical and biological data at Toolik Lake (68°N), McDonald et al. (1996) reported a 3 °C increase in mean July epilimnetic temperatures. Their simulation models have predicted "that with this July temperature increase, YOY lake trout would need to consume >eight-fold more food to achieve the same end-of-year size as historically surviving YOY lake trout" which suggests major food web changes in Arctic Alaskan lakes dominated by lake trout. Based on the Toolik Lake data, Hobbie et al. (1999) have also predicted some profound changes in freshwater ecosystems in the North American Arctic. The effects predicted by them are essentially related to the melting of permafrost which will cause a release of phosphorus into streams and lakes, rendering them more productive. However, they predicted that the continuing warming trends are likely to change the structure of the food web as the top predator (lake trout) in the region becomes extinct as a result of its shrinking habitat.

A number of studies (Osterkamp et al. 2000; Jorgensen et al. 2001; Fedorov and Konstantinov 2003; Gavriliev and Efremov 2003; Osterkamp 2003; Yoshikawa and Hinzman 2003) have shown that long-term permafrost degradation has already started in Arctic areas. These studies show that global warming in the last 30 years has brought soil temperatures to a significantly high level (i.e. 1 to 3 °C warmer than long-term averages). The inevitability of global warming has raised important questions, such as how global warming will impact the Arctic hydrologic cycle and how will this altered cycle impact local and regional atmospheric dynamics, net primary productivity, carbon and methane fluxes, and the flux of river waters to the Arctic Ocean. Permafrost studies of tundra lakes on the western Arctic coast of Canada have suggested that permafrost in littoral terraces of lakes may not be sustainable as a result of changes brought about by global warming (Burn 2002). As such, attempts are being made to predict the impacts of global warming on Arctic watersheds as well as impacts in a pan-Arctic context (Stieglitz et al. 1999; Ling and Zhang 2004). Research has also suggested that "warming of the ground in permafrost regions will result in additional release of carbon to the atmosphere" (Stieglitz et al. 2003), and complex ecosystem responses such as large shifts in functional groups, increased initial decomposition, and increased peat accumulation in boreal peatlands (Camill et al. 2001).

An important consideration with respect to global warming should be its effects on population dynamics, especially in shallow water bodies such as the numerous shallow lakes in the Canadian Arctic and the prairies, where water temperature is similar to the ambient air temperature (Carpenter et al. 1992). Beisner et al. (1997) investigated the effect of temperature (18 °C and 25 °C) on the stability of a common freshwater predator-prey system consisting of *Daphnia pulex* and phytoplankton in different types of mesocosm communities. The results showed that the predator-prey system was destabilized at the higher temperature in all types of communities, and extinction of *D. pulex* always occurred. This effect was enhanced by the addition of a carnivore *Mesostoma ehrenbergii*. Two mechanisms were proposed by these authors for the observed greater population instability at the higher temperature: 1) "the direct effect of temperature on vital rate parameters that destabilized the plant-herbivore interaction *or* 2) an unfeasible plant-herbivore equilibrium with an increase in temperature owing to a shift in the algal community structure to less edible species."

Schindler (2001) has reviewed and discussed the cumulative effects of climate warming and other human stresses on Canadian freshwaters in the new millennium. Supported by *in situ* scientific evidence and predictions of various models, he has provided a number of implications of climate warming on Canadian freshwaters, as follows:

"Climate warming will adversely affect Canadian water quality and water quantity.

"The magnitude and timing of river flows and lake levels and water renewal times will change. In many regions, wetlands will disappear and water tables will decline. Habitats for cold stenothermic organisms will be reduced in small lakes.

"Warmer temperatures will affect fish migrations in some regions. Climate will interact with overexploitation, dams and diversions, habitat destruction, non-native species, and pollution to destroy native freshwater fisheries. Aquatic communities will be restructured as the result of changes to competition, changing life cycles of many organisms, and the invasion of many non-native species. Decreased water renewal will increase eutrophication and enhance many biogeochemical processes. In poorly buffered lakes and streams, climate warming will exacerbate the effects of acid precipitation. Decreases in dissolved organic carbon caused by climate warming and acidification will cause increased penetration of ultraviolet radiation in freshwaters."

Changes in dissolved organic carbon (DOC) are one of the important effects of climate warming (Schindler 2001). Allochthonous DOC is an important determinant of thermocline depth in small boreal lakes, changes related to water clarity, thermal capacity, and longer ice-free periods. Schindler (2001) also forecasts direct effects on fisheries that will include the warming of many unstratified northern lakes to above the optimum temperature for certain species such as lake trout, and thereby lowering the production (presumably due to increased metabolic demands which may not be met through increased food intake). Even at sublethal levels, warming would increase the energy requirements of all fish and, for example, YOY lake trout by "several fold" (McDonald et al. 1996). Also increased thermocline depth will reduce the summer thermal refuges and lake sub-thermocline habitats. Adaptation to elevated temperatures by lake trout has been recorded (Sellers et al. 1998) and in another location these fish species exhibited behavioral adaptation spending much of their time near cold springs and making forays into waters wherein temperatures exceeded usual tolerances in order to feed (Snucins and Gunn 1995).

The effects of climate change as hypothesized by Schindler (2001) above can have far-reaching consequences for freshwater ecosystems, especially in the boreal and Arctic regions (Minns and Moore 1992; McDonald et al. 1996; Hobbie et al. 1999; King et al. 1999; Schindler 2001). It is, therefore, important to consider the effects of climate change and its interaction with other stressors, such as habitat fragmentation, in all conservation and restorative efforts (McCarty 2001).

Relative to regions of lower latitudes, climate warming is much more rapid in the Arctic, hence it is particularly vulnerable (Schindler 2001; ACIA 2004). The earth's climate has already warmed by 0.5 °C over the past century, and recent studies have shown that it is now possible to detect the effects of a changing climate on ecological systems (McCarty 2001). The average Arctic temperature has risen at almost twice the rate as the rest of the world in the past few decades (ACIA 2004). Visual evidence of warming is provided by widespread melting of glaciers and sea ice, and a shortening of the snow season in the Arctic. The environmental implications of climate warming, however, need to be addressed within the context of a number of other changes simultaneously taking place in the Arctic, including the observed increase in chemical contaminants entering the Arctic from other regions, increase in UV radiation due to ozone thinning, over fishing, land-use changes (such as mining and hydroelectric dams) that result in habitat destruction and fragmentation, rapid growth in the human population, and cultural, governance, and economic changes (ACIA 2004).

Ozone thinning and UV radiation: In the last few decades, erosion of earth's protective stratospheric ozone layer has resulted in a significant increase in ultraviolet radiation (UV) reaching the earth's surface, considerably higher in the Arctic than at mid latitudes (Madronich et al. 1995; Kouwenberg et al. 1999*a*, 1999*b*; Waibel et al. 1999). Despite the implementation of the Montreal Protocol (United Nations Environment

Programme (UNEP) 2000*a*), the increase in UV radiation reaching earth's surface has been predicted to continue for decades (World Meteorological Organization (WMO) 2002, 2003; UNEP 2000*b*).

Since the Arctic has long been an area at high risk from UV radiation damage, concerns have been expressed with respect to the environmental impacts of the increasing UV radiation on Arctic organisms and ecosystems (Bothwell et al. 1993; Bothwell et al. 1994; Perin et al. 1995; Waibel et al. 1999; Hessen 2002*a*, 2002*b*; Hessen et al. 2002; Browman and Vetter 2002; Herndl and Obernosterer 2002; Bischof et al. 2002; Häder et al. 2003; Helbling 2003). The combination of biological sensitivity, already-high UV levels and increased ozone depletion in the spring could mean serious challenges for fragile Arctic ecosystems (UNEP 1998).

Literature has appeared in the last couple of decades relating to the effects of UV on aquatic species and ecosystems in higher Arctic latitudes (Hessen 2002*b*). Organisms at the base of the food chain are particularly sensitive to solar UV radiation (Vincent and Belzile 2001). Inhibitory effects of UV radiation on the photosynthetic activity of algae have been demonstrated (Bothwell et al. 1993; Hanelt et al. 1997; Bischof 2000; Brouwer et al. 2000; Lesser et al. 2002). Van De Poll et al. (2002) demonstrated temperature dependence of UV radiation effects in Arctic and temperate isolates of three red macrophytes and Poppe et al. (2003) demonstrated effects of UV radiation on the ultrastructure of several red algae. The latter work provides "insight into the fine structural changes which occur during and after UV exposure and indicate a relationship between the species dependent sensitivity to UV exposure and the depth distribution of different species."

As pointed out elsewhere in this report, bacterioplankton productivity is far greater than previously thought with high division and turnover rates (Fuhrman and Noble 1995). In some oligotrophic freshwater systems (such as the Arctic lakes and streams) bacterioplankton productivity is comparable to or exceeds phytoplankton productivity (Herndl 1997). It is well known that bacterioplankton plays a central role in the carbon flux in aquatic ecosystems by taking up dissolved organic carbon and remineralizing the carbon. This important part of aquatic food web is, however, more prone to UV-B radiation stress than larger planktonic organisms because of a lack of UV screening pigments such as mycosporines or scytonemins (Garcia-Pichel 1994).

A number of studies have demonstrated a variety of impacts of UV radiation on zooplankton, such as UV-B-induced DNA damage and photoenzymatic DNA repair in Antarctic zooplankton (Malloy et al. 1997); UV impacts on grazing activity of *Daphnia* (Van Donk et al. 2001), depth distribution of zooplankton (Leech and Williamson 2001; Hessen et al. 2002) and invertebrate predation (Boeing et al. 2004). Hessen (1993) suggested that UV-induced DNA damage in Arctic zooplankton could be used as a bioindicator of UV radiation. In a comprehensive review of the impact of UV radiation on marine crustacean zooplankton and ichthyoplankton from the estuary and Gulf of St. Lawrence, Browman et al. (2000) concluded that "although exposure to UV can negatively impact crustacean zooplankton and ichthyoplankton populations, these direct effects are likely minimal within the context of all the other environmental factors that produce the very high levels of mortality typically observed in their planktonic early life stages."

Fish larvae and eggs, are vulnerable to increasing UV radiation, especially the pelagic forms and those inhabiting low dissolved organic carbon (DOC) lakes. Kouwenberg et al. (1999*a*, 1999*b*) and Beland et al. (1999) demonstrated susceptibility of cod eggs located at or 10 cm below surface to UV radiation (wavelengths <312 nm) in the Gulf of St. Lawrence. Kouwenberg et al. (1999*a*) suggested that their results "illustrate the relative changes in UV-B impacts that will result from ozone layer depletions expected over the coming decades." However, they also pointed out that "variability in cloud cover, water quality, and vertical distribution and displacement of eggs and larvae within the mixed layer, can all have a greater effect on the flux of UV-B radiation to which eggs are exposed than will ozone layer depletion at these latitudes" (approximately 48°N, 68°W).

The role of DOC in attenuating solar UV radiation is of critical importance in mediating the exposure of freshwater communities to harmful UV radiation (Häder et al. 2003). Kelly et al. (2001) demonstrated the potential of dissolved organic matter to influence benthic colonization through its influence on UV radiation and predicted "that streams with low dissolved organic matter concentrations <5 mg/L could be particularly sensitive to UV radiation." It is also important to note that solar UV radiation has been shown to photolytically degrade DOC into substances such as formaldehyde, acetaldehyde, glyoxylate and pyruvate that are readily taken up by bacterioplankton (Wetzel et al. 1995; Williamson et al. 1999). The increased breakdown of DOC and subsequent uptake of breakdown products by bacteria increases UV-B penetration into the water column. Arctic freshwater and marine ecosystems are often nutrient limited especially with respect to phosphorus and nitrogen (Dohler 1992). Since the uptake of these inorganic nutrients is UV-B sensitive (Dohler 1992), it is likely that this may augment the UV-B sensitivity of Arctic phytoplankton communities. The important role of DOC in the chemistry of fresh waters is well known, where it complexes with heavy metals and makes them less bioavailable to aquatic organisms. Winch et al. (2002) found increased toxicity of a variety of heavy metals to freshwater green alga (*Pseudokirchneriella subcapitata*) in DOC-poor waters. Clearly, all these effects have the potential to alter the biogeochemical cycles in freshwater Arctic ecosystems (Williamson et al. 1999).

Persistent organic pollutants: Over the last few decades an increasing variety and number of persistent organic pollutants (POPs), including organochlorine pesticides, PCBs and mercury have been deposited in the Arctic via atmospheric transport from the more southerly latitudes where these chemicals have been used extensively (Beyer et al. 2000; Muir and Norstrom 2000; Koziol and Pudykiewicz 2001). Many of these chemicals have no source in the Arctic. Volatile compounds are lifted by the atmosphere at lower latitudes and are dumped as the air cools over the Arctic. These compounds accumulate in the Arctic environment within the food chain (Voutsas et al. 2002) and pose threats to aquatic ecosystems and human communities which subsist on fishing and hunting (Larsson et al. 2000; Reid et al. 2000; Fisk et al. 2001; Cullon et al. 2003).

The Canadian Arctic Contaminants Assessment Report (CACAR II) (2003) provides the following trends of POPs in freshwater fish of the Canadian Arctic:

"In contrast to land mammals, levels of mercury are increasing in the organs of some freshwater fish in certain lakes, but there is much variation from lake to lake.

"Mercury levels have increased in the livers of burbot from the Mackenzie River (Dehcho), in some locations by more than 35% since 1985. In some cases mercury levels are above the subsistence consumption level, but all are below the guideline for commercial sale. In other locations, mercury levels are rising and, while not of concern for consumption, should continue to be monitored.

"Loche (burbot) from Yukon lakes do not contain high levels of POPs with the exception of Lake Laberge, where toxaphene levels in livers continue to be high. As with other animals in the North, the new brominated flame retardants are being found in loche (burbot). Levels are very low but are increasing with time and should continue to be monitored.

"Land-locked Arctic charr from Nunavik and Labrador generally contain only low levels of mercury and POPs. In contrast, some land-locked Arctic charr from Resolute Lake on Cornwallis Island contain mercury levels above the guideline for subsistence consumption, though below the commercial guideline. These higher levels may have resulted from some Arctic charr becoming predatory. Levels of POPs are low and not considered to be of concern for human health.

"In the Mackenzie River (Dehcho) Basin and in Nunavut, predatory fish such as lake trout, jackfish (northern pike) and pickerel (walleye) generally contain levels of mercury above both the subsistence and commercial guidelines. In contrast, non-predatory fish such as whitefish generally contain much lower levels. Freshwater fish from Great Slave Lake (Tucho) and Great Bear Lake (Sahtú) contain some of the lowest levels of mercury found in the Canadian North. Levels of POPs are low in all these fish, even in predatory fish, and are not of concern."

It was also reported that mercury levels in lake sediments were increasing and it was suggested that "these increases may be because more mercury is being transported from the south, or possibly because of climate change as mercury is released into the environment from melting permafrost and along with organic matter entering lakes and rivers." It is recommended that there be continued monitoring the trends of POPs and mercury residue in the Arctic ecosystems (CACAR II 2003).

A case study of cumulative watershed effects: Lessons can be learnt from the long-term studies of cumulative watershed effects of forest harvesting in Carnation Creek (British Columbia). The project spans decades and reveals the need for long-term committed research in order to understand events, and biological, chemical and physical processes that occur upon land disturbance; an understanding that is related to the rate of recovery from forest harvesting and related practices and restorative measures.

More than 20 years of intensive studies on the Carnation Creek watershed were conducted to examine the cumulative impacts of forest harvesting. Hartman et al. (1996) provided five generalizations derived from this long-term study of hydrological, fluvial-geomorphological, thermal and trophic processes. These authors stated that, with respect to the watershed: the processes were extremely complex and almost fully interconnected; each species and life stage of salmon and trout responded differently to impacts of forestry activities; responses differed geographically; watershed orientation or topography affected impacts, and some responses occurred ≥4 km downstream from the location of logging impact; responses differed in duration and some persisted for a few years, while others continued longer than originally expected and may continue for centuries; impacts of forestry activities may combine with conditions in the marine environment to limit fish population numbers.

The study showed that landslides and debris torrents caused by logging operations in the Carnation Creek area modified steep-slope tributaries and the mainstem of the creek. In addition, bank erosion altered stream channels on the alluvial flood plain. These effects were cumulative over time and reduced the quality of spawning habitat, and rearing habitat for juvenile salmonids. The most important impacts of forestry activities involved channel changes (i.e. changes in large woody debris dynamics), and changes in temperature regimes in the creek (Hartman et al. 1996). Initial effects involved temperature changes and streamside-related channel structure alterations. For example, stream temperature increased 0.5 °C, 0.75 °C, and 3.2 °C during winter, spring and summer respectively, for the decade following logging of the riparian zone of Carnation Creek (Hartman et al. 1996). While the elevated stream temperature during summer enhanced fish growth, the increased stream temperature during winter and spring caused rapid embryonic development and early emergence of salmonid fry.

The Carnation Creek study revealed that "each activity conducted within an overall forest management plan may affect the physical components of an ecosystem differently" (Hartman and Scrivener 1990). These authors further concluded that "whether these effects had positive or negative impacts on fish and other stream biota depends upon the specific activity conducted, the species present, and the life stage of each species of fish." Thus, in Carnation Creek, "the forestry practices that increased stream insolation, water temperature, and nutrient levels increased the numbers, growth period and size of coho salmon fry (*Oncorhynchus kisutch*), but reduced the marine survival of chum fry (*O. keta*). They also increased the growth period and growth rate of trout fry (*O. mykiss* and *O. clarki*), but the growth decreased among the older age groups of both coho salmon and trout" (Hartman and Scrivener 1990). The study also revealed that the stream-side logging activities decreased the stability of the stream channel and its organic debris, which reduced fish survival and numbers.

Similar cumulative watershed effects of logging were also observed in a long-term study of Casper Creek on the Pacific coast of California (Reid 1998) For a more recent and detailed review of fisheries and forestry interactions see Northcote and Hartman (2004).

Cumulative effects and diamond mining in northern Canada: All diamond mining companies have supplied information to assist with an assessment of the potential impacts of their projects on the environment. This information is contained within their respective environmental impact statements.

Concerns have been expressed with respect to the potential for cumulative environmental effects due to diamond mining activities on the biophysical environment including water quality (Stiff 2001). These concerns were environment based and included the incremental stress placed on the Bathurst caribou herd and grizzly bear and their habitat, reduced ambient air quality and its effects on wildlife and vegetation, and potential cumulative effects on water quality of mine discharges from BHP Billiton and Diavik diamond mines (Stiff 2001). The environmental assessments of BHP Billiton and Diavik were the subject of criticism by several non-government organizations. For example, the Canadian Arctic Resources Committee charged that the consideration of cumulative effects was incomplete because it failed to consider the effects of projects for which information was readily available" (Stiff 2001). Similarly, the Mackenzie Valley Environmental Impact Review Board pointed out that the scope of cumulative effects assessment was too narrow, and that certain projects such as the expansion of the Ekati diamond mine were not included in the assessment (Stiff 2001). An unpublished report to DFO by McCart (Arctic ecologist and retired member of the BHP Billiton Independent Review Agency) concluded that, "particularly for the biological component of the aquatic ecosystem at Lac de Gras, there are likely to be significant changes at the community level, but that, because of the high level of generality of the AEMP, there is a high probability that we will never know" (McCart 2003). Contrasting perhaps,with the opinion of McCart, Hanks and Williams (2000), however, considered that from the perspective of the diamond mining industry there were no (at that time) cumulative effects due to mining, although there were site-specific impacts.

A workshop "to facilitate the identification of cumulative effects indicators that could be incorporated into a cumulative effects monitoring program for the Coppermine River basin" was held in 1999 (MacDonald 1999). Based on the nature and extent of potential cumulative effects and the probability of their occurrence, mining exploration and development, transportation corridors and infrastructure development, municipal development, long range transport of atmospheric pollutants, and climate change were identified as the most important stressors in the Coppermine River basin. The potential effects associated with these stressors identified at the workshop included changes in hydrological conditions, loss of physical habitats, and degraded water and sediment quality conditions. The workshop participants also recognized that a variety of natural factors (such as naturally-elevated mercury levels) in the Coppermine River basin have the potential to exacerbate the effects of stressors caused by human activities and, as such, these natural factors should be considered in the assessment of cumulative effects.

Current initiatives in cumulative effects monitoring and assessment: Several initiatives are underway in response to the need to ensure adequate assessment of cumulative effects in northern areas. Some of these are presented below.

NWT Cumulative Effects Assessment and Management (CEAM) Strategy and Framework: The CEAM Strategy and Framework is a collaborative effort to improve environmental management and stewardship in Canada's north. The process was jointly initiated by the Federal Ministers of Indian and Northern Affairs and the Environment following the comprehensive study for the Diavik Diamonds Project in June 1999. The process was led by a Steering Committee composed of representatives from Aboriginal organizations, industry, environmental and non-governmental organizations, the federal and territorial governments, and the Mackenzie Valley Environmental Impact Review Board. (Up-to-date information on all aspects of the NWT CEAM Strategy and Framework is available at http://www.ceamf.ca). The overall objective of the NWT CEAM Strategy and Framework is to make recommendations to decision makers to facilitate protection of ecological integrity, building of sustainable communities, including social and cultural dimensions, and responsible economic development within a sound environmental management framework.

The CEAM Strategy and Framework was developed by the NWT CEAM Steering Committee, in consultation with a broad range of interested parties including Aboriginal people. A "Blueprint for Implementing the Cumulative Effects Assessment and Management Strategy and Framework in the NWT and its Regions" has been prepared by the NWT CEAM Steering Committee (revised November 2003). It uses a broad definition of environment, which includes social, economic and cultural aspects in addition to natural and biophysical factors.

The Cumulative Impact Monitoring Program (CIMP): The CIMP is an initiative designed to meet requirements of land claim agreements and of the *Mackenzie Valley Resource Management Act* (MVRMA), and to support informed resource management decisions throughout the NWT. When implemented, CIMP is expected to provide resources to help fill gaps in current environmental monitoring, report on the State of the NWT environment and the cumulative impacts of land and water uses and deposits of wastes, and encourage community-based monitoring and capacity building. The audit component of the program will independently evaluate at least once every five years the state of the NWT environment, the effectiveness of CIMP, the effectiveness of land and water regulation in NWT, and make recommendations to improve resource and environmental management.

The Draft Regional Plan of Action for the Slave Geological Province (SGP): The Regional Plan of Action for the SGP is part of the broad-based CEAM initiative and its purpose is to "make recommendations to decision-makers to facilitate the protection of ecological integrity, the building of sustainable communities (including social and economic dimensions), and responsible economic development within a sound environmental management framework." The specific objectives of the SGP include a description of the CEAM Framework for the SGP, provision of a common set of terms and definitions for CEAM, description of the current context in terms of framework components (roles, responsibilities, and accountabilities), description of the links between framework components in the context of the current regime, identification of the gaps in the current regime relative to the framework components, provision of a role in the assessment and management of cumulative effects in Slave Geological Province to Ministers, co-management bodies and others, and engage in consultation, communication and education with a wide range of stakeholders in NWT and Nunavut.

Major ongoing water and sediment quality monitoring programs in the NWT: Northern Rivers Ecosystem Initiative (NREI) by Environment Canada and numerous other agencies since 1998: focus is on pollution prevention, science-based ecological management, resolution of contaminant and nutrient issues, endocrine disruption, long-range transport of air pollution (LRTAP), and continuing environmental research in northern rivers.

Northwest Territories Water Quality Monitoring Program: (Environment Canada since 1960).

Inter-jurisdictional Interim Aquatic Quality Monitoring Program: (Environment Canada and Alberta Environment since 1988, and Government of NWT since 1984) – water and sediment quality data available for Hay and Slave River sites.

Nahanni National Park Reserve Aquatic Quality Monitoring Program: Nahanni National Park Reserve is an Ecological Monitoring and Assessment Network (EMAN) site located in the NWT at the border with the Yukon Territory. Base and precious metal mining occurred upstream of Nahanni prior to park establishment. Nahanni waters, sediments, fish, and caribou have naturally elevated metals levels (Environment Canada and Parks Canada since 1992).

Diamond mine aquatic effects monitoring programs: The diamond mining companies are required to undertake monitoring activities as part of their Territorial Government Water Licences and their *Fisheries Act* authorizations. They are required to report the findings in monitoring reports (monthly Surveillance Network Program water quality reporting, annual Water License reporting, annual Aquatic Effects Monitoring Program reporting) as well as reporting on compensation monitoring (e.g. Panda Diversion Channel Fish Habitat, Snap Lake spawning reef), fish health, and mitigation effectiveness.

The environmental monitoring information generated by the diamond mining industry is available from the following websites:

Concerning BHP Billiton (Ekati), the Independent Environmental Monitoring Agency can be consulted for relevant documents via the following website: http://www.monitoringagency.net/website/Key%20Documents/New%20Key%20documents%20index.htm.

For Diavik information consult the Environmental Monitoring Advisory Board website: http://www.emab.ca/.

Regarding the De Beers Snap Lake Project, the Snap Lake Environmental Monitoring Agency (SLEMA) has recently been set up and is in the process of implementing a library of environmental reports pertinent to the project (contact Louis Azzolini, A/Executive Director, SLEMA, Yellowknife, NWT).

Tahera project documents are available via the following Nunavut Impact Review Board ftp site: http://ftp.nunavut.ca/nirb/Reviews/00MN059%20-%20Jericho%20Diamond%20Mine/167%20Jericho%20Final%20EIS/.

Coppermine Cumulative Effects Monitoring Program (INAC): The Water Resources Division of INAC maintains a network of water quantity and quality stations on the Coppermine River. The department realizes that while this network provides essential information for assessing trends in quantity and quality conditions, it is not comprehensive enough to detect subtle changes in this resource as a result of multiple developments and other ecosystem stresses. As such, INAC has developed a cumulative effects monitoring program for the Coppermine River Basin in conjunction with other federal agencies, universities and Aboriginal organizations. A key component of the program is the continuous monitoring of water quality at the outlet of Lac de Gras. The program includes monthly sampling at 6-8 sites, seasonal sampling at several other sites, and continuous monitoring at the outlet of Lac de Gras. Forthcoming studies are expected to include snow surveys in a small representative basin to develop snowmelt runoff and basin rainfall-runoff ratios, and monitoring of river ice breakup. The program has been included in INAC's 2003/2004 to 2005/2006 Business Plan.

Slave River Environmental Quality Monitoring Program (INAC): Water, suspended solids, and fish quality are monitored at Fort Smith to address transboundary issues. A 5-year program is underway to determine temporal changes in monitoring parameters.

Liard River Environmental Quality Monitoring Program (INAC): Water, suspended solids and fish quality are monitored via this program.

Ecological Monitoring and Assessment Network: Water and sediment quality are monitored at selected sites in the NWT.

COMMENT

It is clear from the review of the scientific literature that the Canadian Arctic presents somewhat unique habitats for fish. This is so because of its extreme biogeochemical and climatic conditions (e.g. extreme cold and extensive ice cover of lakes and rivers, extreme variations in daylight, short growing season, low nutrient supply and productivity, and low biodiversity). Many Arctic lakes typically have a high degree of autonomy and freedom from human influence (Johnson 2002). They are relatively simple ecosystems and the abundance of fish is generally high and comparable to that in lakes further south with much higher productivity.

The results of the literature review suggested that the Canadian Arctic is experiencing environmental stress caused by a variety of factors including economic development (e.g. mining, oil and gas development, and hydroelectric dams), and global changes (e.g. climate warming, ozone thinning, and long-range atmospheric transport of toxic chemicals). Scientific evidence is also mounting to suggest that the Arctic freshwater systems are vulnerable to climate warming. As such, long-term studies should be designed in this context.

It is also clear that the processes that determine the physical state of the atmospheric, terrestrial, and aquatic systems in the Arctic are intrinsically inter-linked and, as such, changes in any of these systems can impact the others (Kerr 1996; Nelson 2003; ACIA 2004).

Biological and ecological aspects

A number of new concepts in freshwater ecology have emerged with respect to the structure and function of freshwater ecosystems over the previous two decades. Most of these concepts have emerged from freshwater research in temperate zones, and two of these are considered to be particularly important in Arctic freshwater ecosystems. That is, freshwater ecosystems are regulated through both "bottom up" resource limitation (i.e. availability of nutrients such as phosphorus), but also through "top down" cascading along the food chain. Substantial evidence has been provided to suggest that microbial food webs ("microbial loop") composed of autotrophic picoplankton, protozoans, and microzooplankton play an important role in the overall trophic dynamics of lakes. It has been demonstrated that in oligotrophic lakes (such as the tundra lakes in the Northwest Territories), the microbial loop and bacterioplankton are responsible for a larger share of the overall production and energy processes of the pelagic community than in more productive lakes in relatively southern latitudes (Stockner and Shortreed 1989; Berman 1990; Weisse et al. 1990; Weisse and MacIsaac 2000).

The limited information from the Canadian Arctic suggests that physical loss of habitat (e.g. destruction of a tundra lake) has the distinct potential to disrupt fish populations and communities, as well as the structure and functioning of ecosystems over both temporal and spatial scales. As such, the impact of such habitat destruction or loss should be evaluated by taking into account all factors that govern both structural and functional aspects of the ecosystem. The comparative qualitative assessment of information on the species of fish selected for this review revealed the relative deficiencies in knowledge between the Arctic environment and the warmer southerly locations. Although information deficiencies will always exist, the paucity of knowledge about even the most important species in the Arctic environment precludes all but very approximate estimates of the impact of developments and compensation for habitat change.

Early life stages of fish are especially vulnerable to both biological and physical changes in habitat. Availability of the appropriate food supply (which can be disrupted as a result of human developments) for various stages in the life cycle of a fish species is critical for survival. Since a number of fish species migrate long distances within the watershed for spawning, over wintering, and feeding, any physical changes within the watershed can disrupt these migrations and hence the survival of those species. Spawning substrate for most species is gravel, cobble, rock, and occasionally, sand and silt. While some species have been demonstrated to select alternative spawning sites and substrates, others (such as Arctic grayling) show considerable fidelity for both. Destruction of such habitats could, therefore, be detrimental to populations utilizing these habitats. Availability of nursery areas close to the spawning sites and the availability of appropriate-size food items can be critical for survival and recruitment. Arctic lakes and streams have extremely low biodiversity; the food chains are short. In some cases, a trophic level is represented by a single species.

Physical disruptions and loss of fish habitat are known to cause changes in turbidity and sedimentation regimes. These changes are often detrimental to various life stages of fish, especially eggs. Physical changes brought about by mining or other human interventions are known to increase turbidity and reduce dissolved oxygen levels (Johnson 1997) and can also mobilize toxic metals which have been shown to negatively impact fish and fish habitat (Kelly 1998).

Recommendations and gaps in knowledge

There is a general understanding that "no branch of environmental science can progress very far without drawing on knowledge from other branches" (National Research Council, NRC 2001, 2002). Accordingly, it is desirable to adopt an ecosystem (holistic) approach to managing the Arctic environment, which recognizes that all aspects of the environment are connected, and, similarly all ecosystems are connected in a functional hierarchy of systems (Keith 1994; O'Riordan 1995; Khan 1997; Jones and Taylor 1999; Cairns 1999; Minns 1999; Schindler 2001; Vorosmarty et al. 2001; Schlosser et al. 2003).

The ecosystem approach, therefore, represents a strategy for the management of all aspects and components of the environment (land, water, atmosphere, and living resources). Such an integrative approach would promote conservation and sustainable use of natural resources in an equitable way. The approach by its very nature would be integrative and collaborative and offer opportunities to foster a common understanding of issues among public, government, and industrial sectors (management, science, and social levels).

Based on the literature that was reviewed, there is a general paucity of information with respect to environmental aspects of the Canadian Arctic, especially concerning habitats necessary for the survival of fishes. Little work has been done on the biology and ecology of fish in the Canadian Arctic, especially with respect to habitat use by various life stages (i.e. alevins, larvae, juveniles, etc.) of fish. Similarly, information is generally lacking on fish survival over winter, that is considered to be a critical and limiting phase for fish production. Such knowledge is essential to understand the implications of habitat change and is particularly important in habitat compensation, remediation and restoration plans (Bradbury et al. 2001).

The autonomous Arctic lakes provide opportunities for research rarely available elsewhere (Johnson 2002). Here, the simply-structured pristine ecosystems provide the opportunity to examine habitat-fish linkages (limnology-fish ecology), and to apply such learning for the betterment of fisheries and habitat (environmental) management. Some information on Arctic fishes is available from the baseline work done for the Mackenzie Valley pipeline and work done in Alaska (Sly et al. 2001). For some species such as lake trout and Arctic grayling, the biological understanding is much better than for other Arctic species (Reist 1997). Because of these deficiencies, information from more southerly locations such as the Great Lakes has been utilized in the assessment of the impacts of developments on fish in the Arctic. But this is problematic for it is not known whether Arctic fish populations and stocks are genetically different from southern populations and consequently have different environmental requirements and tolerances. It has been accepted that the dearth of scientific information is the single most practical problem in the management of fisheries and fish habitat in the Canadian Arctic (Power 1997; Reist 1997; MacDonald 1999; MacDonald et al. 1999). This is a deduction that we endorse after conducting this review.

Because of the uniqueness and harshness of the Arctic environment (i.e. high latitude harsh and cold climate, poor nutrient supply, low productivity, and low biodiversity), Arctic populations of fish are faced with relatively different challenges for survival than those in more southerly regions. Much of the information that is necessary to determine the critical factors for fish survival in the Arctic is essentially missing in the literature.

Our review of the biology of selected species of Arctic fishes revealed several gaps in knowledge which require further research for a better understanding of the freshwater ecosystems of the Canadian Arctic. These areas of research are listed below.

Fish and fish habitat: In this review of the biology of selected species of Arctic fishes, several areas of information were identified that require research. These include the location of spawning grounds; habitat requirements for spawning (including fidelity for spawning sites and substrates) and frequency of spawning; routes and timing of migration; standing crops in specific watersheds; post-emergent dispersal of fry and migration to nursery and over-wintering areas; habitat requirements for over wintering; residence time in spawning, rearing, summer feeding, and over-wintering areas; species interactions for food and space; species interactions and trophic dynamics (including predator-prey relationships) in stream and lake fish communities; factors (biotic and abiotic) controlling fish species distribution in lakes and streams; data to enable determination and delimitation of the basic units of biological management or stocks, as well as the biological structure (such as age, size, etc. of those exploited) and their temporal and spatial dynamics (Reist 1997); physiological and behavioral adaptations of fish to Arctic conditions; factors controlling food availability for various life stages of fish; the role and significance of the "microbial loop" in energy flow within oligotrophic freshwater systems in the Canadian Arctic; environmental factors controlling recruitment; and detailed studies of species association, competition and predation; and resilience of fish species to the direct and indirect effects of human interventions.

Specific Recommendations: The groundwater upwelling areas in Arctic rivers and streams which provide over-wintering refugia for juvenile and resident fish are considered critical habitats for a variety of fish and need to be identified and protected (Cunjak 1996; Alfredsen and Tesaker 2002; Morse and Hicks 2005).

Because substrate size has been demonstrated to be important to escape predation for certain species such as lake trout (Biga et al. 1998), it is necessary to include and evaluate this critical habitat feature in relation to considerations of compensation and restoration.

Difficulties with ageing lake trout have been reported by many authors (Scott and Crossman 1973; Healey 1978; Martin and Olver 1980). As such, it is an impediment to adequately determining age at maturity. Therefore, there is great danger in comparing age at maturity and other parameters based on ages determined using different techniques. More consistent and precise techniques are required to age fish and further work is required to better understand the factors controlling age at maturity in Arctic fish populations.

In the Canadian Arctic (Scott and Crossman 1973; Power 1997) and Alaska (Morrow 1980), lake trout populations in some lakes are reported to migrate into streams in summer for feeding, and return to lakes to over winter. But information is lacking with respect to the extent and nature of the use of stream habitat for spawning and feeding by lake trout. These aspects, therefore, need to be investigated.

Some evidence has been provided to show that during the summer feeding period, Arctic grayling in Alaskan streams adopt a larger-older-fish-upstream distribution pattern (Hughes and Reynolds 1994; Hughes 1998*b*). However, the mechanisms involved in the maintenance of these population processes are poorly documented (Hughes 1999). Although several interpretations have been provided (Tack 1980), this distribution pattern in streams adds to the complexity of migrations and habitat selection by Arctic grayling. Further work is required to elucidate this complex situation.

Fidelity to over-wintering habitat sites in Arctic grayling has been strongly suggested by the studies reported by West et al. (1992). Over-wintering refugia need to be considered critical habitat for the survival of Arctic grayling and further work is required with respect to the maintenance of optimum environmental conditions (e.g. dissolved oxygen, turbidity) in these areas.

After attaining sexual maturity Arctic grayling are reported to usually spawn each year, but examples exist where this is not so (Scott and Crossman 1973; Falk et al. 1982). The causal factors for this behavior need to be investigated.

The role of ice cover on survival of eggs of lake whitefish requires further investigation under Arctic tundra conditions as there is contradictory evidence in the literature (i.e. Miller (1956) reported significant mortality in lake whitefish eggs, but Freeburg et al. (1990) reported a nine-fold increase in egg survival under ice cover in two subsequent years).

Hatched fry of lake whitefish are reported to either move downstream in smaller streams or remain in shallow inshore lake areas adjacent to spawning grounds for about two months (Morrow 1980). Since these habitats are critical for survival (recruitment) their significance in the tundra landscape needs to be investigated.

As there have been reports of fidelity to spawning sites and substrates by lake whitefish in Ontario lakes (Anras et al. 1999*a*, 1999*b*), this aspect needs to be investigated in the relatively smaller and shallower Arctic lakes.

Divergence in the growth pattern of lake whitefish in similar lakes has been demonstrated (Bidgood 1973), and has been attributed to the presence or absence of predators. This aspect of predator-prey relationships needs to be investigated for fish in lakes and streams being impacted by human interventions such as diamond mining.

Although adult burbot are usually found in relatively low-velocity areas along lake bottoms (Ford et al. 1995), they are also reported to tolerate high summer peak flows in the glacial Tanana River in Alaska (Breeser et al. 1988). With such variations reported, there is a need for further research to determine optimum current velocity for burbot and for other species in areas where habitat manipulation might involve changes in current velocities.

Although adult burbot are reported to prefer turbid waters (Chen 1969; Hatfield et al. 1972, both cited in McPhail 1997), little is known about the turbidity levels which could be harmful to them. In fact, it has also been reported that they are more abundant in the silty waters of the Yukon and Tanana Rivers than in smaller and clearer tributaries (Ford et al. 1995). This aspect requires further investigation to establish the optimum turbidity requirement for burbot habitat and the behavioral issues surrounding the use of such habitats which may include the use of turbid waters as cover to assist food capture by this ambush predator.

Several authors have concluded that consumption of lake trout eggs by sculpins could be a significant source of lake trout mortality (Biga et al. 1998; Hudson et al. 1995; Chotkowski and Marsden 1999; Fitzsimons et al. 2002) and deduced that predation by sculpins on salmonid eggs could be critical for the recruitment of the latter. Since in many Arctic tundra lakes, lake trout and slimy sculpin are sympatric (Martin 2001), it would be important to understand the implications of sculpin predation on lake trout eggs on recruitment.

Relative to regions of lower latitudes, climate warming is much more rapid in the Arctic, hence Arctic ecosystems are particularly vulnerable to global warming (Schindler 2001; ACIA 2004). The environmental implications of climate warming need to be addressed within the context of a number of other changes simultaneously taking place in the Arctic, including the observed increase in chemical contaminants entering the Arctic from other regions, increase in UV radiation due to ozone thinning, over fishing, land-use changes (such as mining and hydroelectric dams) that result in habitat destruction and fragmentation, rapid growth in the human population, and cultural, governance, and economic changes (ACIA 2004).

The state of knowledge in the Coppermine River drainage: There are a number of gaps in knowledge that are critical in the assessment of environmental conditions and temporal trends in the Arctic. Although reviews of the ambient environmental conditions in the Coppermine River watershed suggest that the "Coppermine River and its tributaries are currently in a relatively pristine state" (MacDonald et al. 1999; MacDonald 1999), concerns have been raised with respect to the cumulative effects of mining and other human activities on the fisheries resources of this watershed. The review by MacDonald et al. (1999) concluded that the data on various aspects of the watershed are generally insufficient for an adequate understanding of the Coppermine River ecosystem. A concerted effort is, therefore, required to acquire data on climate, hydrology, and on the structure and functioning of the ecosystem comprising the Coppermine drainage. In particular, the following areas require further attention, as identified by MacDonald et al. (1999).

Although the long-term climatic data are available for the lower (or northern part) of the Coppermine watershed in the vicinity of Kuglukuk, the data for central and southern portions of the basin are lacking (MacDonald et al. 1999).

Data on hydrology are essentially limited to 8 locations in the Coppermine River drainage, which is insufficient to describe the hydrological conditions in the watershed, especially the long-term temporal trends (MacDonald et al. 1999).

A comprehensive and systematic water and sediment quality program has not been carried out on a watershed basis in the Coppermine River drainage. Although water quality data are available for the past four decades, these data are insufficient for evaluating the seasonal or long-term spatial trends in water quality in the drainage. Long-term monitoring undertaken by the diamond mining companies will assist understanding of any cumulative effects due to such developments. However, according to MacDonald et al. (1999), the limitations of information include a lack of data from many of the sub-basins, lack of seasonal variability data, inconsistent analytical methodologies, and detection limits that are often greater than water quality guidelines. Data on sediment quality are especially inadequate. Long-term, watershed-wide water and sediment quality data are needed from areas of watersheds known to have metal enrichment (MacDonald et al. 1999).

Data on phytoplankton, periphyton, aquatic macrophytes, zooplankton, and benthic invertebrate communities are limited, especially in the connecting streams or channels between lakes. Data on the watershed and long-term trends are prominently lacking. Although, the data describing the diversity of fish species in the basin are available, data on the structure and function of fish communities are lacking (MacDonald et al. 1999).

The need for long-term research: In order to fill the gaps in knowledge, a directed and long-term commitment with clearly-stated science and management objectives will be required. The importance of such long-term and large-scale ecological research has been demonstrated in Canada at the Experimental Lakes Area (Schindler et al. 1985) and in the United States through the Long Term Ecological Research (LTER) Network (Hobbie 2003; Symstad et al. 2003; Kratz et al. 1991; Kratz et al. 2003; see commentary by Levings in Consultations, Samis et al. 2005). As suggested by Reist (1997), the specific needs of such an effort should be: "(1) development of comprehensive workable models for fisheries management, (2) development of associated research plans, (3) prioritization (and proper sequencing) of the information needs (components) of such plans, and (4) implementation of research effort."

INDUSTRIES

Details of the history and operation of the diamond, oil sands and placer gold mining industries are provided to give a brief perspective on these activities. The level of information provided differs among the mining sectors described and relates not only to the longevity of the operations, but also to the nature of the impacts on fish habitat. For example, because of the ubiquitous release of sediment from human developments, and its potentially negative effects on fish habitat, this subject has been emphasized in the report within the placer mining section. These comments are applicable to the other mining sectors whose operations can also result in sediment releases into fish habitat. The main regions in Canada where these activities are occurring are identified in Figure 3.

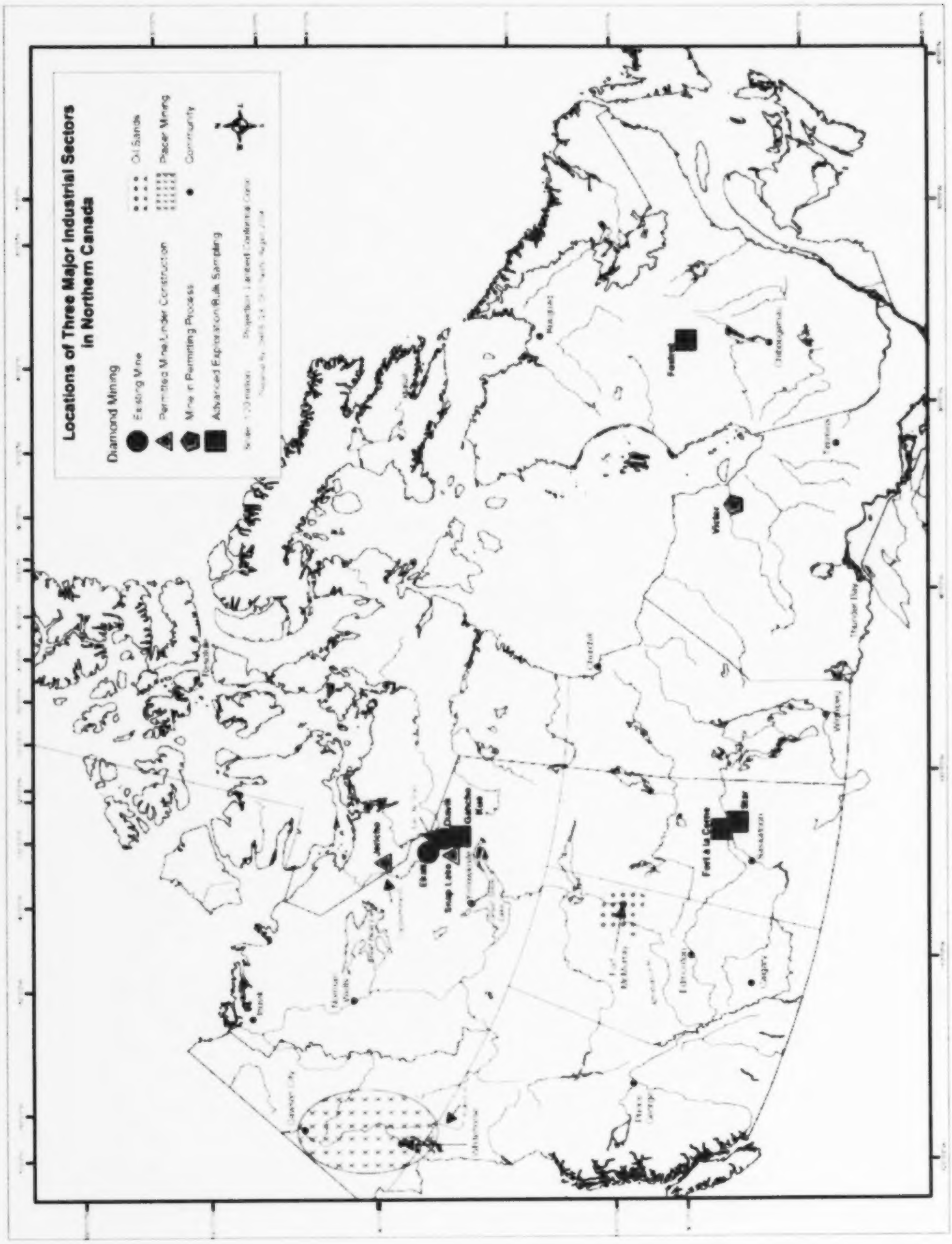

Locations of Three Major Industrial Sectors
in Northern Canada
Diamond Mining
Existing Mine
Permitted Mine Under Construction
Mine in Permitting Process
Advanced Exploration Bulk Sampling
Oil Sands
Placer Mining
Community
Jericho
Ekati
Diavik
Gahcho Kue
Snap Lake
Fort à la Corne
Star
Victor
Thunder Bay
Winnipeg
Churchill
Edmonton
Calgary
Saskatoon
Fort McMurray
Dawson City
Whitehorse
Resolute

Figure 3. Map showing the main locations of current and proposed diamond, oil sands and placer gold mining operations in Canada.

DIAMOND MINING IN CANADA

Location

Diamonds are found in kimberlite, which, according to the American Geological Institute (1997) is “an ultramaficigneous rock consisting of >35% olivine, with a groundmass containing olivine, monticellite, plogopite, spinel, carbonate, serpentine and apatite.” Precambrian shields around the world often contain kimberlites, and Canada has the largest shield area in the world. Within the Canadian Shield, the Lac de Gras area, 320 km northeast of Yellowknife in the Northwest Territories, has over 250 kimberlite intrusions (pipes and dykes). The pipes are of small diameter (typically 75 to 300 m). Eighty percent of these kimberlite pipes in the Lac de Gras area are under small lakes and 20% outcrop, or are under surficial materials (Kjarsgaard 2003).

The diamond exploration technique widely used in Canada and other recently-glaciated areas relies on an indicator mineral survey technique: “As glaciers advance, they erode, homogenize and re-distribute the components of the bedrock that they pass over. If tracers of diamondiferous kimberlite are found in the re-distributed glacial material, their bedrock source must lie in the up-ice (analogous to upstream) direction” (Saskatchewan Industry and Resources 2005).

Historical perspective

Golder Associates Ltd. (2003) reported that, “In 2002, Canada led the world in diamond exploration, having spent US $47 million.” This expansion was revealed in an article in the Ottawa Citizen (2004) where it was noted that, apart from the existing and proposed mines in the Northwest Territories, Nunavut and Ontario: Ekati, Diavik, Snap Lake, Jericho, and Victor, there is widespread diamond exploration activity in Nunavut, Quebec, Ontario, Saskatchewan and Alberta (refer to Figures 3 and 4).

The production of diamonds was forecast to grow and, according to Santarossa (2004), “recent production data suggest that by the end of 2003 Canada will have produced almost 15% of the world’s supply of diamonds….” In 2004, Canada was confirmed as the third largest producer of diamonds (by value), behind Botswana and Russia (Bruna Santarossa, Statistical Economist, Minerals and Mining Statistics Division, Natural Resources Canada, Ottawa, Ontario; pers. comm.).

Methods, environmental setting, and scope

BHP Billiton – Ekati: From the early 1980s, prospector, Chuck Fipke from Kelowna, BC, and his partner, Stu Blusson sampled for minerals associated with the occurrence of diamonds. In 1985, the trail of indicator materials focused in an area 320 km northeast of Yellowknife, in the region of Lac de Gras. By fall 1989, Fipke and Blusson’s company, Dia Met, had staked nearly 160,000 ha near Lac de Gras. In August 1990, BHP and Dia Met agreed to joint operating terms (Frolick 1999).

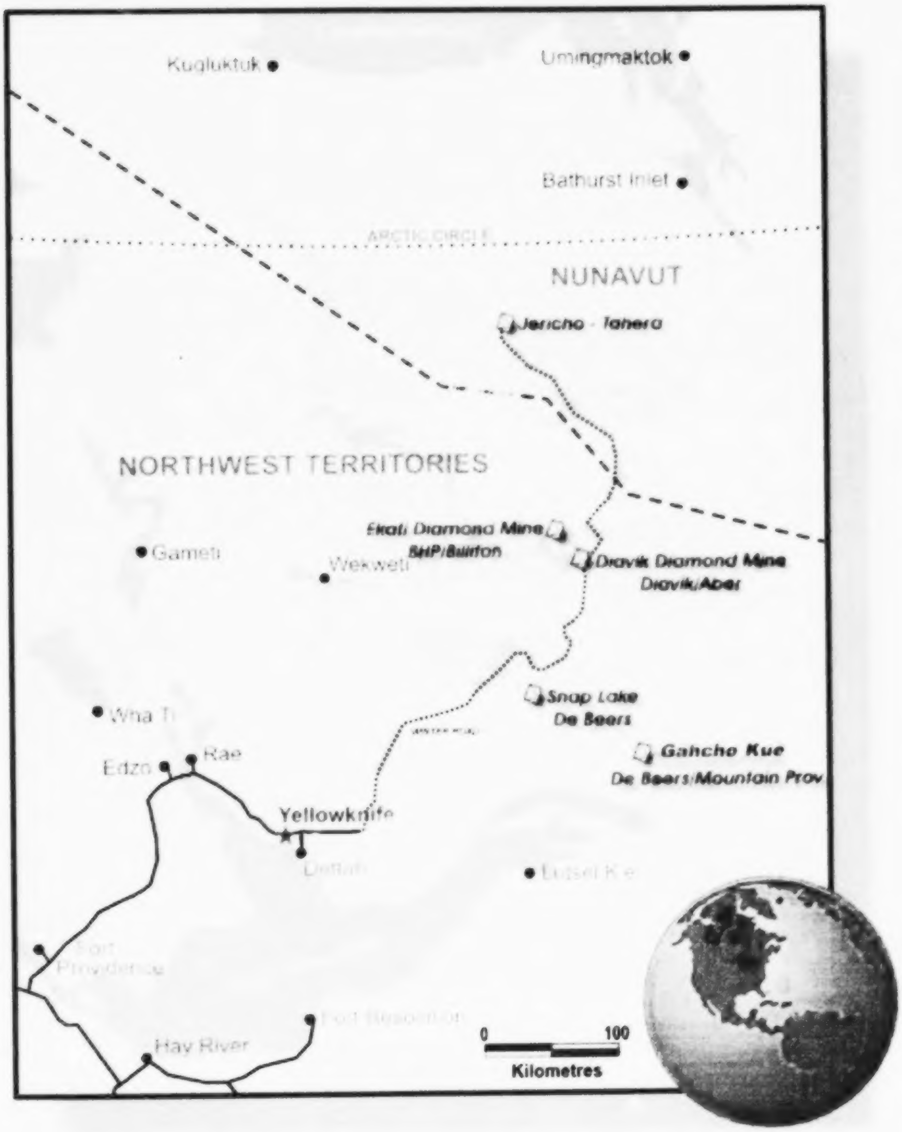

Kugluktuk
Umingmaktok
Bathurst Inlet
ARCTIC CIRCLE
NUNAVUT
Jericho - Tahera
NORTHWEST TERRITORIES
Ekati Diamond Mine
BHP/Billiton
Diavik Diamond Mine
Diavik/Aber
Gameti
Wekweti
Snap Lake
De Beers
Gahcho Kue
De Beers/Mountain Prov
Wha Ti
Edzo
Rae
Yellowknife
Dettah
Lutsel K'e
Fort
Providence
Fort Resolution
Hay River
0
100
Kilometres

Figure 4. Location of existing (Ekati and Diavik) and planned (Snap Lake, Jericho, and Gahcho Kué) diamond mines in the Northwest Territories and Nunavut (adapted from Diavik Diamond Mines Inc.).

In September 1991, BHP drilled into the bed of Point Lake in the Lac de Gras region, and verified that a kimberlite pipe lay beneath at a depth of about 120 m (Frolick 1999). The announcement of this discovery started the biggest staking rush in Canadian history (Saskatchewan Industry and Resources 2005).

By spring 1992, BHP/Dia Met had staked 400,000 ha. De Beers Canada had claims of over 4,000,000 ha (10 million acres) (Frolick 1999). Other companies, including Aber, followed suit and within days of the initial announcement, BHP/Dia Met's claims were surrounded by other claims (Frolick 1999). It has been estimated that over 16 million hectares were staked over an 18-month period following the September 1991 Point Lake kimberlite discovery (Excell 2003*a*).

In July 1994, Minister Copps (Environment Canada) convened an Environmental Assessment Review Process (EARP) Panel to consider the application by BHP to mine diamonds. On November 1, 1996, Minister Irwin (Indian and Northern Affairs Canada) issued federal Cabinet approval for BHP's Ekati mine (Frolick 1999).

According to the January 7, 1997 authorization issued to this diamond mining company, "To compensate for the loss of fish habitat associated with the Project, BHP Minerals shall implement the Fish Habitat Compensation Agreement negotiated between the Department of Fisheries and Oceans (DFO) and BHP Minerals." In accordance with this authorization which recognized that "The Project will impact directly on 12 lakes within the claims block," BHP placed $1.5 million into a Habitat Compensation Fund. Six lakes (Panda, Misery, Koala, Fox 1, Alexis and Leslie) were to be dewatered to provide access to, and enable the exploitation of underlying kimberlite pipes; one (Airstrip) was to be dewatered for access to granular resources for construction; four (Long, Brandy, Willy and Nancy) were to be filled with process plant tails; and one (West Panda) was to be covered by a waste rock dump. In addition, a number of interconnecting and some ephemeral headwater streams associated with the above-named lakes were to be diverted. Harmful alteration of stream habitat was to be compensated for through "fish habitat creation and enhancement" within a channel constructed to divert water from Panda Lake to Kodiak Lake (the Panda Diversion Channel, PDC).

Frolick (1999) reported, "BHP's plan was to develop the five best diamondiferous pipes commencing with open pit mining of Panda, followed by both open pit and underground mining of Misery, Koala, Fox and Sable, which, with better grades than Leslie, had replaced that pipe. The projected life of the mine was 25 years." In June 2001, the Ekati mine was an 80-10-10% joint venture between BHP Billiton (BHPB), and geologists, Chuck Fipke and Stu Blusson.

In order to enable the BHP Diamond's Inc. Misery Pit project, DFO issued two *Fisheries Act* section 35(2) authorizations – one for King Pond and King-Cujo Streams (September 21, 2000) and another for Desperation Pond and Desperation-Carrie Stream (April 4, 2002). On January 17, 2003, DFO issued an additional authorization for the destruction of Sable, Two-Rock, and Beartooth Lakes, Pigeon and Big Reynolds Ponds, and reaches of Pigeon Stream, thereby enabling BHPB's Sable, Beartooth and Pigeon mining project.

As of May 26, 2003, the Ekati claim block comprised about 344,000 ha, with 150 known kimberlite pipes, 8 of which were in the mine plan. Approximately one-third of the Ekati claim block is covered by water (Excell 2003*b*).

It has been reported (Mining Technology 2005) that, to date, 314 ha of tundra habitat have been used for construction of the mine, and 611 ha of the total lease area of 10,960 ha has been affected by the Ekati operation. Hanks and Williams (2000) stated that there are site-specific effects due to mining or mines currently being developed but there was no regional cumulative effects attributable to exploration or mining (comments made in 1999). Rehabilitation of the affected areas is reported to be an on-going, long-term aim (Mining Technology 2005).

At BHPB's site there are currently 6 open pit mines in various stages of stripping or mining (i.e. Panda, Koala, North Koala, Misery, Fox, Beartooth), with two more planned (i.e. Pigeon, Sable) (J. Dahl, Area Chief, Habitat Management, DFO, Yellowknife, NWT; pers. comm.). Figures 5 and 6 show diamond mine pits in the Arctic.

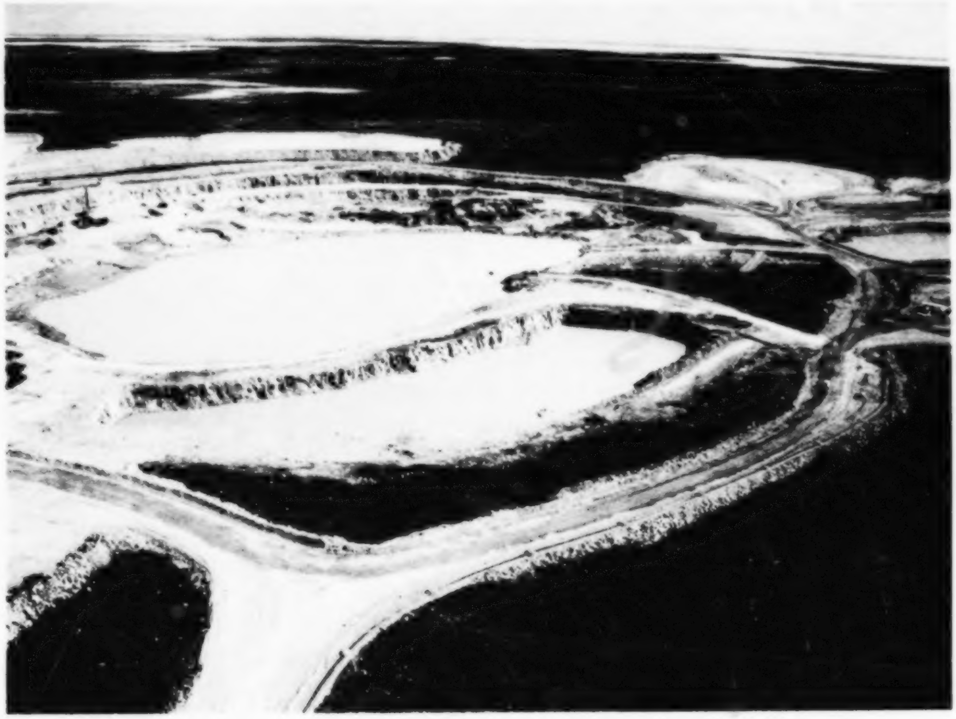


Figure 5. Lake being de-watered for access to diamond ore in the Arctic.

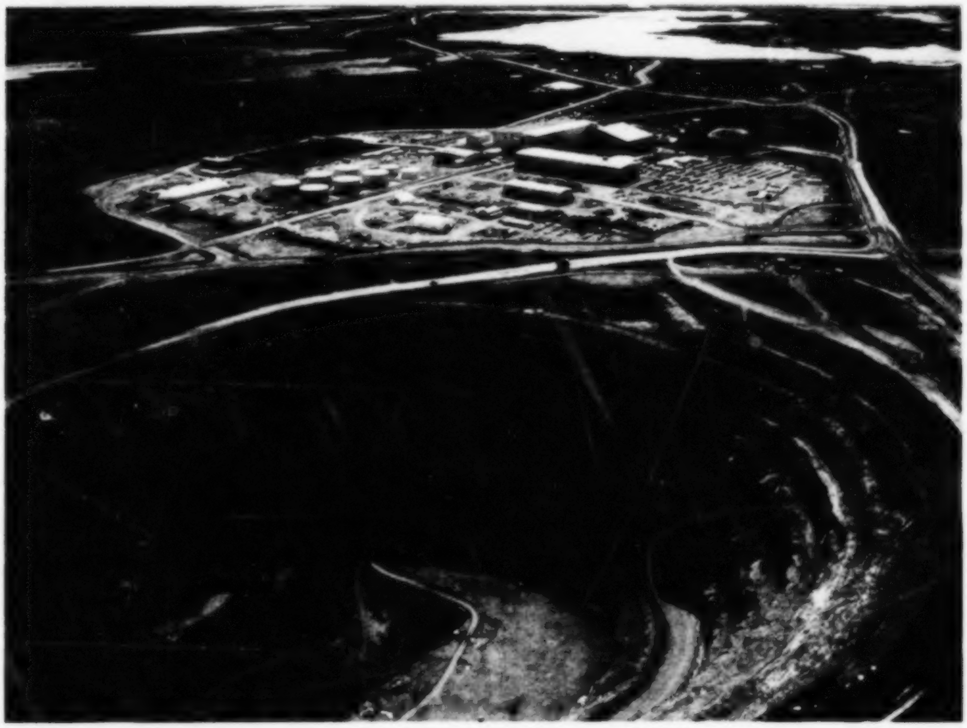


Figure 6. An open diamond mine pit in the Arctic with the mill in the background: a lake was eliminated to permit the extraction of kimberlite.

Diavik Diamond Mines Inc.: The Diavik Diamond Mine is an unincorporated joint venture between Diavik Diamond Mines Inc. (60%) and Aber Diamond Mines Ltd. (40%), a wholly-owned subsidiary of Aber Diamond Corporation of Toronto, Ontario.

Aber found diamond-bearing kimberlite pipes in the Lac de Gras area in 1991 and staked areas east of BHP/Dia Met claims thereby commencing the Diavik diamond mining project (Government of the Northwest Territories 2005).

The Diavik project is centred on 20-km^2 East Island in Lac de Gras, 300 km northeast of the Yellowknife (Diavik Diamond Mines Inc. 2005). The Diavik mine is 30 km southeast of the Ekati mine, and intrudes into Lac de Gras.

The Diavik Diamonds project received regulatory and investor approval to build the mine by December 2000 (Diavik Diamond Mines Inc. 2005).

DFO issued Diavik a *Fisheries Act* section 35(2) authorization that permitted damage to fish habitat (August 2, 2000 to December 31, 2025). This enabled open pit and underground mining of kimberlite pipes at the company's Lac de Gras site. Six lakes on East Island (named e1, e3, e6, e7, e8, e10) were to be destroyed during mine development, along with streams, and areas of Lac de Gras for dyke construction and de-watering, open pit mining, dredging, jetty construction, and mine water disposal and management (North Inlet). Fish habitat compensation to achieve NNL is to involve enhancement of a lake on East Island and a stream on the adjacent West Island, and include development of spawning, rearing and shoreline habitat and other features in and around the dyked areas of Lac de Gras. The development of the shallow water habitat within the dykes is to be carried out upon completion of mining in each open pit.

Diavik's production started in January 2003. Mining is to be focused on the four Lac de Gras/East Island pipes (starting with A154S and A154N; then moving to A418 and A21) (J. Dahl, Area Chief, Habitat Management, DFO, Yellowknife, NWT; pers. comm.). In a November 19, 2003 News Release (Aber Diamond Corporation 2003), it was announced that Diavik Diamond Mines has presented an updated mine plan, "…to expand production while increasing ore reserves through better definition of the existing resources and through aggressive exploration of the 240 thousand hectare Diavik Property."

Exploration activities of most interest to fish habitat managers are winter drilling programs to identify kimberlite targets, and the large-diameter drill programs to obtain bulk samples for an analysis of their economic potential (drilling programs are often intensive, involving numerous drill holes to identify and delineate the kimberlite targets (J. Dahl, Area Chief, Habitat Management, DFO, Yellowknife, NWT; pers. comm.).

Snap Lake, De Beers Canada: Winspear Resources undertook initial exploratory activities on claims at Snap Lake (NWT). On January 24, 2000, DFO issued Winspear Resources a *Fisheries Act* authorization for destruction of Lake IL1 and associated Stream S30. In 2000, the property was acquired from Winspear by the South African diamond-mining company De Beers. The authorization was amended on January 15, 2001 and June 13, 2001 to reflect changes in company ownership and to amend the valid authorization period. The purpose of the authorization was to enable the development of a processed kimberlite containment (PKC) facility (tailings impoundment area) for the advanced exploration phase of the Snap Lake Mine.

De Beers submitted its project description for the permitting of a mine at Snap Lake in February 2001. The proposed mine site is located about 220 km northeast of Yellowknife and has an expected life span of 20 years (Government of the Northwest Territories 2005). De Beers has one kimberlite at Snap Lake. Unlike most other kimberlite intrusions, the Snap Lake kimberlite is a sheet 2- to 3-m thick, extending from the surface and beneath Snap Lake for about 1 km. No lakes need to be drained to access this kimberlite (J. Dahl, Area Chief, Habitat Management, DFO, Yellowknife, NWT; pers. comm.).

On October 15, 2003, De Beers announced that the environmental assessment for the Snap Lake Diamond Project was approved by the Minister of Indian and Northern Affairs. The company also stated that the project would enter into the regulatory phase with the application for permits from the Mackenzie Valley Land and Water Board (MVLWB). De Beers received a water licence from the MVLWB on May 31, 2004. A *Fisheries Act* authorization was issued on August 30, 2004 for HADDs in Stream S29 due to the placement of a sediment pond berm, and in Snap Lake associated with placement of rock for the mine water intake and outlet, and for the zone of turbulence created by the operation of the treated effluent diffuser.

Mine construction is anticipated in 2005, with full production in 2007 (De Beers Canada 2005). The project "footprint" is described as being less than 550 ha; mining is to be by underground methods (De Beers Canada 2003).

Jericho, Tahera Diamond Corporation: The environmental assessment process for the proposed Jericho diamond mine, located in Nunavut, has been completed. On January 21, 2003, Tahera Diamond Corporation submitted its "Final Environmental Impact Statement" for the proposed mine to the Nunavut Impact Review Board (NIRB) (S. Critch, Habitat Management Biologist, DFO, Iqaluit, Nunavut; pers. comm.). Tahera plans to construct and operate a diamond mine near the north end of Contwoyto Lake in West Kitikmeot. Operations would involve an open pit mine with the discharge of processed kimberlite (tailings) into Long Lake (9.2 ha), and two (0.9 and 0.7 ha) ponds. With current reserves, the company expects the mine and processing plant to have an 8-year life-span (Tahera Diamond Corporation 2005).

The mine footprint is expected to be 221.8 ha. No lake will be drained to access the ore, but a stream is to be diverted, and a lake and two ponds will be eliminated due to the storage of processed kimberlite (tailings) (J. Dahl, Area Chief, Habitat Management, DFO, Yellowknife, NWT; pers. comm.).

In February 2004, the NIRB recommended to the Minister of Indian and Northern Affairs Canada that the project be allowed to proceed to the regulatory phase, provided that certain terms and conditions are met by the proponent. Tahera would require several *Fisheries Act* section 35(2) authorizations because it plans to use Long Lake and two adjacent ponds for processed kimberlite disposal, to construct a causeway in Carat Lake, and to divert Stream C1 around the mine pit. It is anticipated that there will be impacts on the outlet streams due to lowered water quality from the processed kimberlite containment area, altered stream flow, and loss of stream length. In addition, concern has been raised over the effects of blasting on fish and their eggs (S. Critch, Habitat Management Biologist, DFO, Iqaluit, Nunavut; pers. comm.).

Efforts to fully mitigate impacts to fish habitat have been unsuccessful, thus fish habitat compensation, pursuant to the Habitat Policy, will be required to offset losses. The environmental assessment of the project was completed in June 2004 and the NIRB project certificate was finalized in mid July 2004. The project proceeded to the regulatory phase in the fall of 2004 and concluded with the issuance of a water license by the Nunavut Water Board (D. Moggy, Fish Habitat Biologist, DFO, Iqaluit, Nunavut; pers. comm.). DFO issued an authorization to Benachee Resources Inc. (a wholly-owned subsidiary of Tahera) on April 15, 2005.

Victor Diamond Project, De Beers Canada: De Beers Canada Exploration Inc. has identified a diamond resource, approximately 90 km west of the First Nation community of Attawapiskat, within the James Bay lowlands of Ontario. The resource consists of two kimberlite pipes, referred to as Victor Main and Victor Southwest. The proposed development is called the Victor Diamond Project. Construction would be for a 3-year period starting in the winter of 2006, followed by a 12-year mine life, and a 5-year reclamation phase, with most reclamation occurring in the first 2 years of this period.

The project impacts on fish and fish habitat include the realignment of 2.5 km of South Granny Creek, complete destruction of several muskeg ponds, dredging within the Attawapiskat River and the dewatering of the Nayshkootayaow River, the North River, two tributaries of the Nayshkootayaow River, and potentially the Lawashi River. With proposed mitigation and compensatory measures, the predicted impacts on the fisheries are not likely to be significant. The comprehensive environmental assessment was expected to be completed by June 2005 (T. Hoggarth, Habitat Impact Assessment Biologist, DFO, Parry Sound, ON; pers. comm.).

Gahcho Kué (Kennady Lake), DeBeers Canada: De Beers Canada conducted an exploratory drilling program during winter 2003/04 and collected baseline data in support of an application for full-scale mine development at Kennady Lake, NWT. De Beers has interests in three kimberlite pipes at this site (i.e. 5304 N and S; Hearne N and S; Tuzo), and preliminary plans indicate the need for the dyking off and dewatering of several bays of Kennady Lake for potential open pit mining (J. Dahl, Area Chief, Habitat Management, DFO, Yellowknife, NWT; pers. comm.). De Beers was to make a decision to proceed which would be followed by the submission of an application for a water license in 2005.

Regulatory and governance issues associated with diamond mining

DFO administers the fish habitat protection provisions of the *Fisheries Act.* Section 35(1) of the *Fisheries Act* prohibits the carrying out of any work or undertaking that results in the harmful alteration, disruption or destruction (HADD) of fish habitat. Section 35(2) enables the Minister to authorize HADDs.

Environment Canada (EC) has the lead administrative authority for section 36 of the *Fisheries Act* (the pollution prevention section), including associated regulation development, administration and enforcement. DFO's Minister is accountable to Parliament for the whole of the *Fisheries Act.*

In contrast to diamond mining, pursuant to section 36 of the *Fisheries Act*, the Metal Mining Effluent Regulations (MMER) preclude the use of natural, fish-frequented water bodies as tailings impoundment areas for metal mines unless they are listed on Schedule 2 of MMER. Thus, the regulations would have to be amended by Cabinet if any new natural, fish-frequented water bodies were to be listed as tailings impoundment areas for metal mines (MMER specifically do not apply to diamond mines). Since 1997, 5 lakes have been eliminated for metal mining.

Management of waste rock, tailings and pit water in permafrost regions presents unique challenges that are not generally encountered in the more temperate southern climatic regions. Due to the risk posed by melting and erosion, excavation of lagoons for waste water management and tailings disposal is atypical in permafrost areas. Damming of valleys or drainage areas to create tailings impoundments has been undertaken, however, this approach does not typically eliminate impacts on lakes which may be numerous in many northern "valleys." Using section 35(2), DFO officials have authorized whole lake destruction through the deposition of tailings from diamond mines.

Fisheries and Oceans Canada has enabled the progress of the diamond mining industry by authorizing the elimination or partial destruction of 26 lakes since 1997 for this sector. The elimination of lakes is escalating in relation to the needs of the mining sector in Canada.

There are regulatory and operational differences in how tailings are managed in diamond mining versus metal mining. In June 2003, Environment Canada held the first multi-stakeholder workshop on diamond mining in Canada. Aboriginal groups, government, industry and environmental non-government organizations were present. Golder Associates Ltd. (2003), Environment Canada's workshop consultant, was to provide information on the current regulatory requirements for diamond mining.

In 2003, at the annual Premiers' conference (Charlottetown, July 9-11), participants gave recognition to the growing importance of the diamond mining industry in Canada and "directed their Ministers to develop an action plan for a national diamond strategy that maximizes the benefits to Canadians from all stages of the emerging diamond industry and to report back to Premiers. Premiers invited the federal government and other stakeholders to join the provinces and territories in the development of this important national initiative." (Government of Canada 2003). In August 2004, this action plan for the national diamond strategy was published (Council of the Federation 2004).

Given the growing prominence of diamond mining in Canada, it is incumbent upon regulatory agencies to ensure the availability of appropriate information in order to make sound and consistent decisions. Current decision making on the harmful alteration, disruption or destruction (HADD) of fish habitat, and associated fish habitat compensation (to achieve NNL of the productive capacity of fish habitat) is hampered by a paucity of scientific knowledge about critical habitat requirements and functional linkages among them for the life cycle stages of northern fish species, and the efficacy of fish habitat restorative and compensatory measures.

DFO Habitat Managers and other practitioners require a policy framework to guide rigorous and consistent decision making. There have been no watershed-level assessments with respect to diamond mining, and decisions to authorize the loss and degradation of lakes have been made independently. Limited fish habitat compensation has been attempted owing, in part, to community resistance to the enhancement of the productivity of pristine aquatic systems. Accordingly, there is a need for better understanding of fish habitat requirements in order to compensate for the elimination of lakes, their inherent biota, and their contribution within larger functional ecosystems. These management needs can be addressed through the provision of new and focused information about habitats that are critical for fish species and assemblages and for ecosystem functioning.

DFO has developed an Environmental Process Modernization Plan (EPMP), which includes a risk management approach to implementing the fish habitat protection provisions of the *Fisheries Act* (an outline of the EPMP is presented later in this report in the section entitled "New Management Stategies"). This approach, which includes a Risk Management Framework (RMF) and Pathways of Effects (POE) model, maps the potential effects of a wide range of industrial activities on fish and fish habitat. It also improves predictability and coherence, and includes a new governance model. Applying approved guidelines and best management practices within the model improves the effectiveness and efficiency of the referral review process for applications for authorizations under section 35(2) of the *Fisheries Act*.

The Pathways of Effects model is an approach that relies upon quantitative scientific input to identify potential impacts of activities on fish and their habitat. The validity of its use is related to the quality of the information upon which it relies. Therefore, through its use, and evaluation, information requirements will be identified.

OIL SANDS MINING

Location

There are four major reserves of oil sands in Alberta – Peace River, Wabasca, Cold Lake, and Athabasca (refer to Figures 3 and 7). Regarding surface oil sands mining, in addition to existing mines at Syncrude Canada Limited, Suncor Energy, and Albian Sands Energy, new mines have been approved for Fort Hills Energy (demonstration plant), Shell Canada (Jackpine) and Canadian Natural Resources Limited (Horizon, under construction). In the application stage are Suncor (Voyageur), Shell's Albian Sands (Muskeg River Mine Expansion) and Imperial Oil (Kearl Lake) – all in northeastern Alberta. Several additional oil sands mining proposals are pending (J. Brownlee, Section Leader, Alberta Energy and Utilities Board, Fort McMurray, AB).

Ninety per cent of Canada's oil sands lie deep below the surface and cannot be recovered by surface (open pit) mining techniques. *In situ* processes are used to extract some of the deposits. No single method of *in situ* (deep deposit oil sands) recovery can be applied to all oil sands deposits, since the bitumen varies considerably among and within deposits. The Athabasca deposit is the only oil sands deposit with reserves shallow enough to be mined from the surface (J. Shamess, Impact Assessment Biologist, DFO, Edmonton, AB; pers. comm.). Photographs of oil sands operations are provided in Figures 8, 9, and 10).

Covering an area of 27,500 km^2 (2,750,000 ha), the Athabasca Deposit is the world's largest mineable oil deposit, containing some 800 billion barrels of bitumen (Cumulative Environmental Management Association, CEMA 2000). The deposit has three layers of oil-bearing sand, one above the other, separated by beds of silt, sand and shale. The oil sands are covered by overburden up to 760 m deep, comprising muskeg, glacial tills, sandstones and shales. About 7% of the Athabasca deposit lies under less than 76 m of overburden, making it accessible to surface mining techniques.

Historical perspective

The oil sands occur in layers at different depths in each deposit. About 100 million years ago, the surface of most of Alberta and Saskatchewan consisted of a series of ridges running northwest to southeast. Streams flowing from the Rocky Mountains on the west, and from the Precambrian Shield on the east, brought sand and shale to fill in the hollows between the ridges. Later, this area was the location of the inland seas that distributed the sands more widely. Finally, they were overlain with thick deposits of shale. How the sands became saturated with oil is unknown. One reason for the uncertainty is that bitumen exhibits the characteristics of both young and old petroleum, which have an absence of lighter fractions, relatively high thermal activity and the presence of sulphur, nitrogen and oxygen-containing compounds. Fresh water would have removed the more soluble lighter hydrocarbons and provided oxygen that enabled bacteria to further degrade the bitumen and its lighter fractions.

According to Alberta Community Development (2005), the first large scale commercial operation, Great Canadian Oil Sands (now Suncor Energy), introduced German manufacturer O&K bucketwheels from the coal mining industry when it opened in 1967. Syncrude opened in 1978 and introduced gigantic draglines.

It is projected that over the period of oil sands mining expansion (1997-2025), that new revenues to the Alberta and federal governments will be worth $200 billion (Suncor Energy 2004).

Methods, environmental setting, and scope

Alberta oil sands are hydrophilic. The sands are bonded firmly together by grain to grain contact. Each grain of sand is covered by a film of water, which is surrounded by a slick of heavy oil (bitumen – a thick form of crude oil that only flows when heated or diluted with lighter hydrocarbons). Bitumen makes up 10 to 12% of the oil sands in Alberta, with mineral matter, sands and clays making up the remainder.

It has been theorized that over time light crude oil was transformed by the actions of water and bacteria into a highly viscous bitumen – at a concentration of between 1 to 20% in sand (Alberta Community Development 2005). "About 80% of the oil sands in Alberta are buried too deep below the surface for open pit mining. Accordingly, this oil must be recovered by *in situ* techniques. Using drilling technology, steam is injected into the deposit to heat the oil sand, thereby lowering the viscosity of the bitumen. The hot bitumen migrates towards producing wells, bringing it to the surface, while the sand is left in place…." (Alberta Community Development 2005). Challenges facing the *in situ* process include the management of water used to make steam for the abstraction process (Alberta Community Development 2005).

Before mining begins, the muskeg and overburden which covers the oil sands is removed. The muskeg is saved for reclamation while the overburden (a thick layer of clay, silt and gravel) is used to build dykes for tailings ponds, or deposited in piles called overburden dumps.

Oil sands are mined using electric or hydraulic shovels with buckets that can hold up to 100 tonnes, and are capable of filling hauler trucks with a capacity of 240-360 tonnes. The operating mines deliver about 0.45 million tonnes of oil sands per day to ore-preparation plants (Globe and Mail 2005). In 2004, the combined operations of Syncrude, Suncor, and Albian mined an average of 1.1 million tonnes/day of oil sands. This amount of oil sands mined is equivalent to 598,772 barrels of synthetic crude oil per day (Alexandra MacEachern, Communications and Community Relations Advisor, Alberta Energy and Utilities Board; pers. comm.).

Two types of extraction of oil from the oil sands occur. They involve "conditioning drums," and "hydro transport pipelines" respectively (refer to Siemens Westinghouse 2005).

Crushers and sizers in the ore-preparation plants prepare the oil sands ore prior to mixing it into a hot water slurry for delivery to the conditioning drums or to the hydro transport pipeline. The hydro transport system allows the bitumen to begin to separate from the sands as the slurry is pumped through the lines, thus less energy is required for the hot-water extraction process.

Next, the bitumen is separated from the sand in "separation cells." The separation cells permit the bitumen to float to the surface and the sands and some fines to settle out. In a secondary extraction process, fine clays are removed, after which the bitumen "froth" is diluted with naphtha and processed to remove minerals and water. The resultant solution is stored in holding tanks before being processed further.

The water, clay, sand and residual bitumen (termed coarse tailings) are pumped to holding ponds where they may be transferred directly to tailings ponds or, treated with the consolidated tailings before being pumped to settlings ponds. Coarse tailings are deposited into a settling pond to segregate the coarser material such as sands from the finer material which forms a fine tailings mixture. The consolidated tailings process is designed to facilitate the separation of water from the sands and clays in tailings. Impacts from these large-scale oil sands mining activities relate not only to changes in the physical landscape and watercourses but also to the potential chemical contamination of waters.

Alberta's oil sands, which are estimated to contain 174 billion barrels of remaining established crude bitumen (the estimated recoverable volume), also contain in-place reserves of a minimum of 1.6 trillion barrels (McCrank 2003). They are one-third of the world's known oil reserves (Alberta Community Development 2005), and occupy an area of about 140,800 square kilometres (McCrank 2003). Oil sands currently represent about 52.7% of Alberta's total production of oil, and about 34.8% of all oil produced in Canada. Oil sands production is expected to represent 50% of Canada's crude oil output, and 10% of North American production (Alberta Department of Energy 2004); the oil sands deposits are large enough to supply total world needs for up to 15 years (Alberta Community Development 2004).

Regulatory and governance issues associated with oil sands mining

Oil sands mining affects fish habitat through the destruction of watercourses and riparian areas, as well by impacting water quality through the generation and release of contaminants. The former issues fall under section 35 of the *Fisheries Act* and accordingly are administered by DFO, whereas the latter are managed by EC under section 36.

A number of different organizations have been developed to focus on the collection and management of information associated with the effects of oil sands mining on humans and the environment.

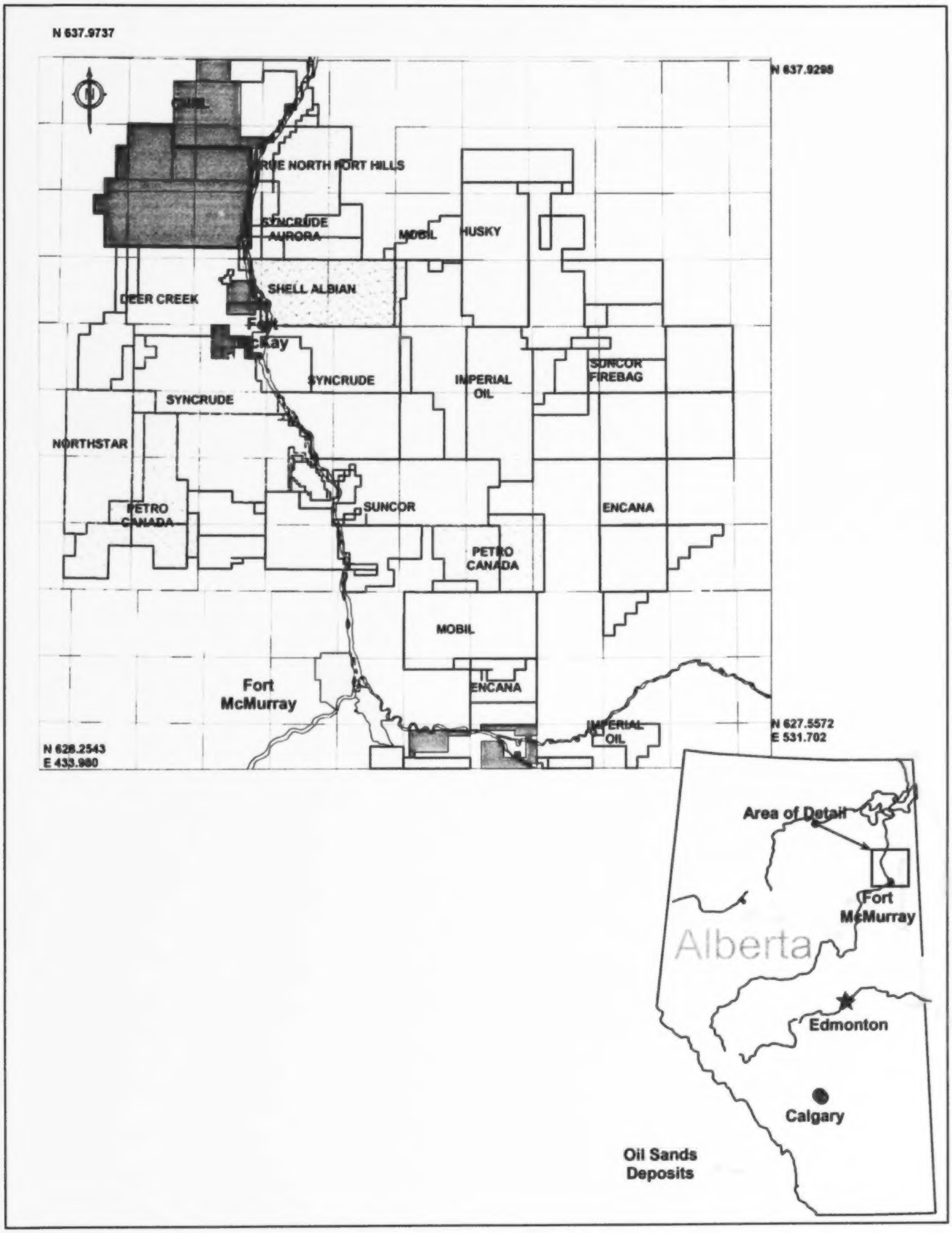
Cumulative Environmental Management Association (CEMA) is a multi-stakeholder initiative in northeastern Alberta established in 2000 to manage the cumulative environmental impacts of industrial development in the Wood Buffalo region of Alberta. CEMA is a collaborative group that is to develop environmental objectives and management plans to address priority environmental issues.

CEMA's mandate is to make recommendations on how to best manage cumulative impacts and protect the environment in the region. These recommendations will be based on scientifically-founded limits and will use information from existing research as well as traditional knowledge. Recommendations from CEMA will be referred to the appropriate regulatory agency for implementation. CEMA will follow up on all recommendations.

CEMA uses consensus-based decision making at all levels. This process results in scientifically-based recommendations that are acceptable to all CEMA members.

The technical and scientific work on developing management recommendations to address priority issues is done through specialized technical working groups (under the direction of the members). Several of these working groups are divided into task groups, based on issues. The working groups are accountable to the members by way of their terms of reference and annual work plans and budgets.

In addition CEMA has established a number of standing committees to assist the organization in core functional areas (e.g. communications). These committees are accountable to the CEMA membership (J. Shamess, Impact Assessment Biologist, DFO, Edmonton, AB; pers. comm.).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Alberta
Edmonton
Calgary
Oil Sands Deposits

Figure 7. Location of Alberta oil sands developments (courtesy of CNRL).

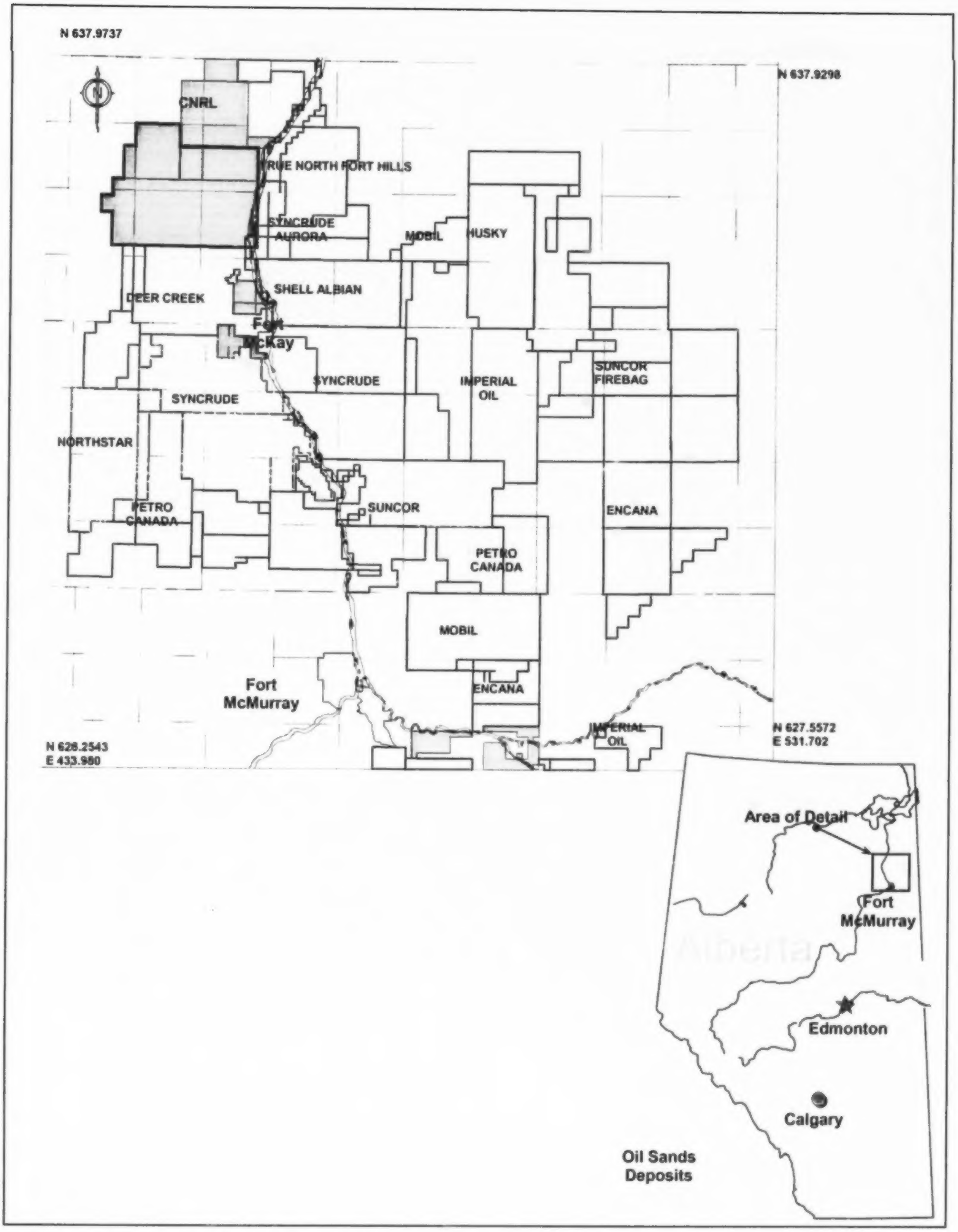
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Figure 7. Location of Alberta oil sands developments (courtesy of CNRL).

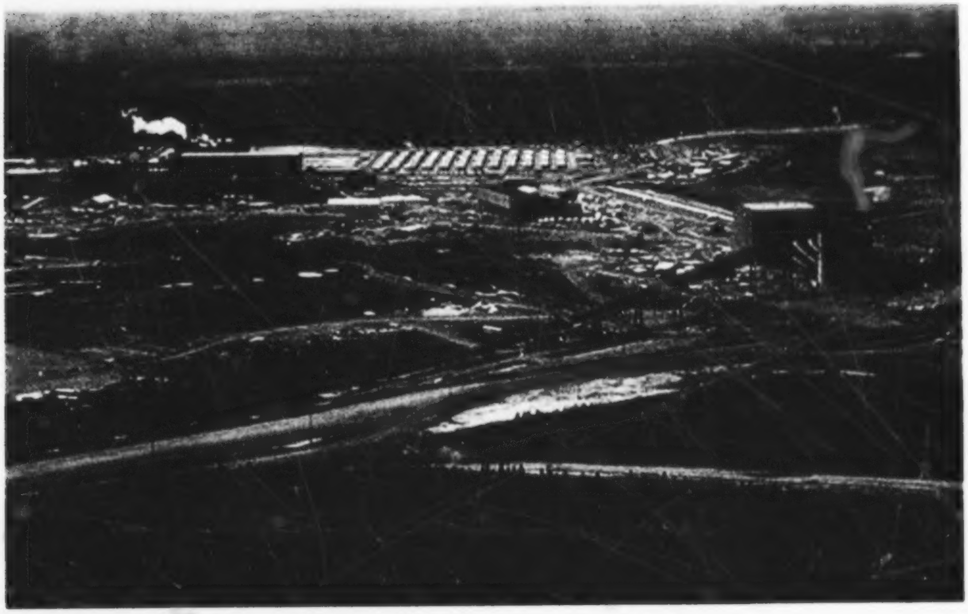


Figure 8. Aerial photograph of an oil sands process plant (courtesy of J. Shamess).

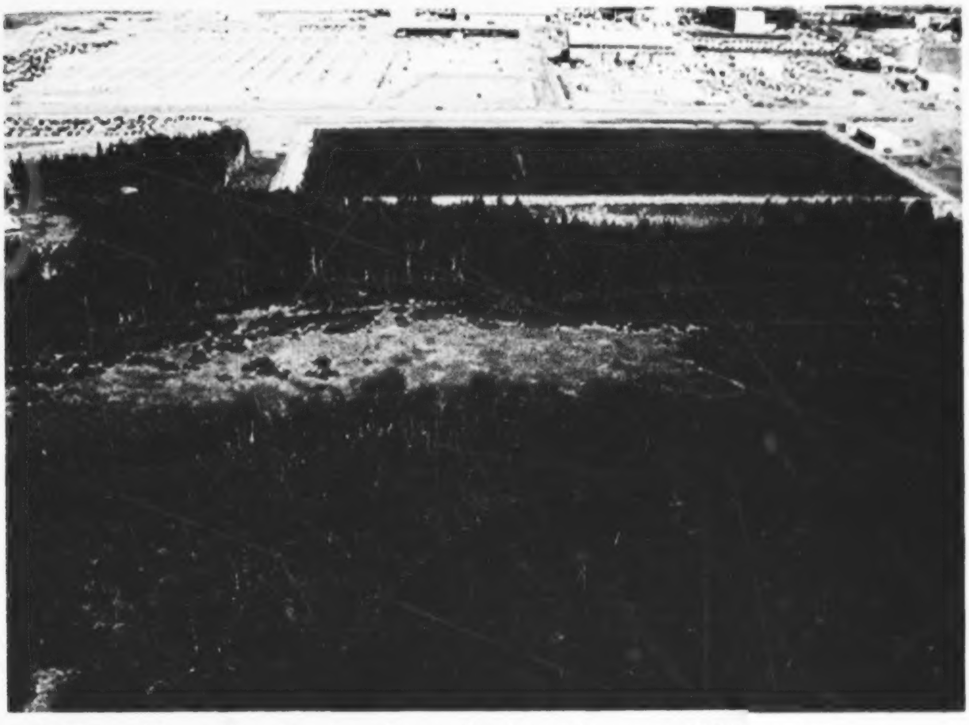


Figure 9. Muskeg River adjacent to oil sands operational pond (courtesy of J. Shamess).

Figure 10. Aerial view of oil sands plant complex (courtesy of J. Shamess).

PLACER GOLD MINING (Yukon)

Much of the historical perspective presented below was abstracted from publications that were prepared for government, the placer mining industry and the public. These documents were prepared in response to the need (identified by the Yukon Placer Mining Implementation Review Committee (IRC)) to investigate the impacts of placer mining in the Yukon Territory (Norecol Environmental Consultants Ltd. 1989; Seakem Group Ltd. 1992; a report to the Minister of Fisheries and Oceans (Implementation Steering Committee, ISC 2004); and unpublished data of DFO).

Location

Placer gold mining occurs in many locations throughout the Yukon Territory but it is primarily focused in regions within the extensive Yukon River basin (refer to Figures 3 and 11).

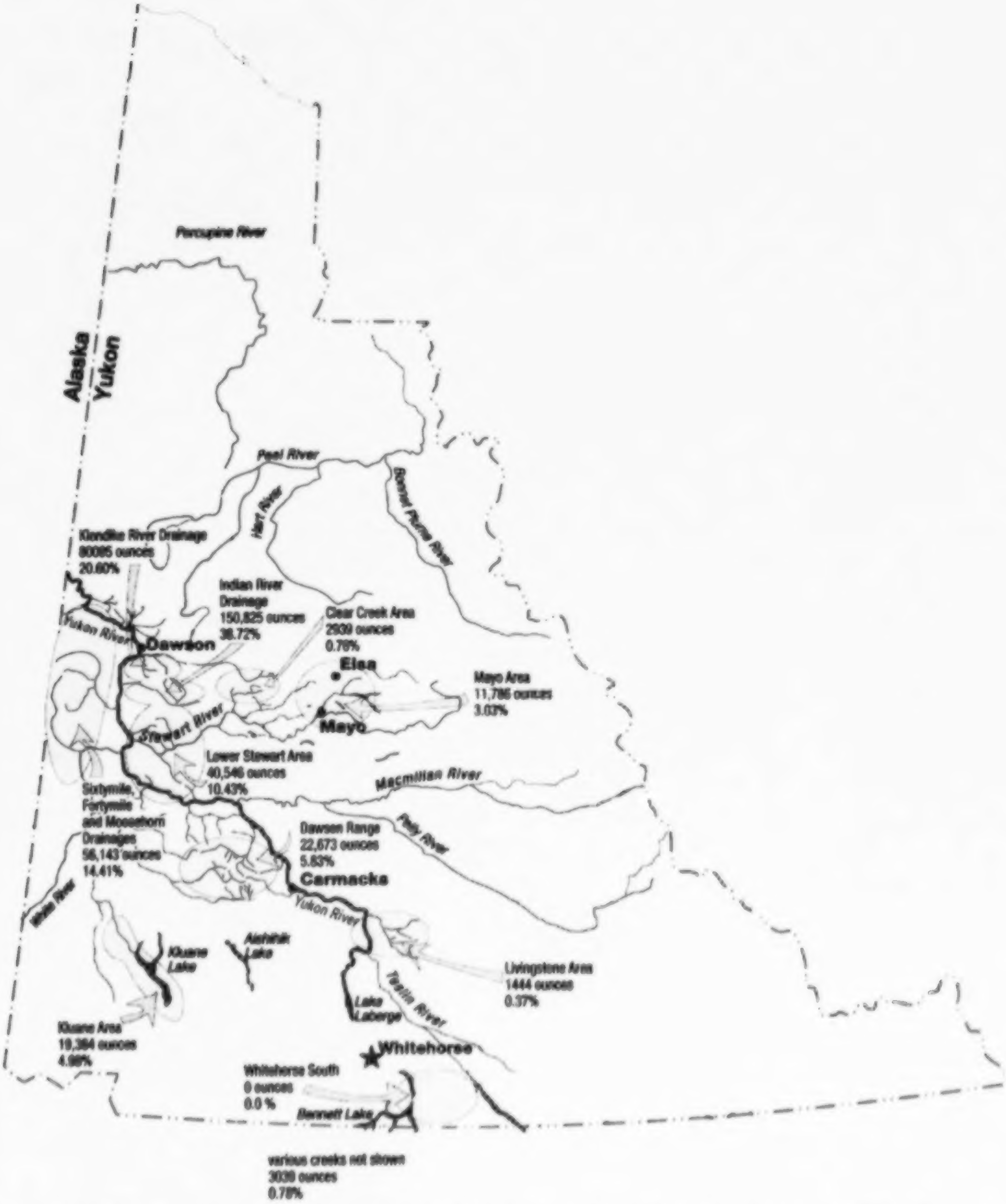

Porcupine River
Alaska
Yukon
Peel River
Hart River
Bonnet Plume River
Klondike River Drainage
80085 ounces
20.60%
Indian River
Drainage
150,825 ounces
38.72%
Clear Creek Area
2939 ounces
0.76%
Yukon River
Dawson
Elsa
Mayo Area
11,786 ounces
3.03%
Mayo
Stewart River
Lower Stewart Area
40,546 ounces
10.43%
Macmillan River
Sixtymile,
Fortymile
and Moosehorn
Drainages
56,143 ounces
14.41%
Dawson Range
22,673 ounces
5.83%
Pelly River
Carmacks
White River
Yukon River
Kluane
Lake
Aishihik
Lake
Livingstone Area
1444 ounces
0.37%
Teslin River
Lake
Laberge
Kluane Area
19,384 ounces
4.98%
Whitehorse
Whitehorse South
0 ounces
0.0 %
Bennett Lake
various creeks not shown
3030 ounces
0.78%

Figure 11. Placer gold mining in regions of the Yukon (adapted from Indian and Northern Affairs Canada 2003).

Most of the gold has been extracted from unglaciated areas of the Yukon Plateau in portions of the Klondike, Sixtymile, Lower Stewart and Indian River drainage basins (Seakem Group Ltd. 1992). However, tributaries of the Stewart River in the Mayo district and others in the Whitehorse area also have produced gold (Norecol Environmental Consultants Ltd. 1989).

Historical perspective

The earliest finds of gold in the Yukon occurred in the mid 1800s but it was not until the major discovery on Bonanza Creek in 1896 that placer gold mining became a significant activity that generated over one million ounces per year in the early 1900s. Thereafter the production declined in relation to diminishing high quality reserves of gold. The price of gold had remained constant at US$20/oz until 1934 when the US government set the price at US$35/oz. This action stimulated more mining and production doubled from that in the early 1930s to about 100,000 oz annually.

While the price of gold remained constant, production declined despite a Government of Canada subsidy that was introduced in the late 1940s, and by the late 1970s annual production was under 10,000 oz.

A revival of mining in size and scope occurred in the 1970s when the price of gold was deregulated and accordingly the price escalated. In 1980 the price of gold exceeded US$800/oz, but by 1990 it had decreased to under US$400/oz. However, production increased in the late 1980s to over 100,000 oz annually which, at that time, was the highest production level since the 1950s.

The extent of exploration and mining activity is related to the price of gold and, at present, the annual production is approximately 100,000 oz (<$40 million). In 2002 placer mining was 2.3% of the Yukon's GDP (ISC 2004). The current royalty rate is $0.375/oz exported gold (i.e.< about $37,500 based on 100,000 oz).

The total recorded fine gold production from 1885 to date is about 12.5 million ounces, valued at US$4.4 billion at today's prices (ISC 2004).

Methods, environmental setting, and scope

Methods: Placer gold is typically extracted from its surrounding substrates by processes that include washing or sluicing large volumes of earth and retaining the heavier metal during the washing process. The exposure of likely placer gold deposits requires the removal of "overburden" (vegetation and related soils, frozen or not). In all situations, the land is disturbed to extract the gold, through the construction of roads and associated infrastructure, and also by the removal of material in and adjacent to watercourses.

Because of the proximity of placer mining to watercourses, the activities typically impact upon fish habitat. These impacts may be from the physical disturbance of the watercourses themselves and the surrounding landscape, and also from the discharge of sediments from sluicing and other operations (see Figures 12 and 13). Thus, primary concerns for the management of fish habitat relate to the effects on aquatic organisms of elevated levels of sediment deposited and suspended in watercourses, the modification and stability of these watercourses, and the adequacy of compensatory and restorative measures.

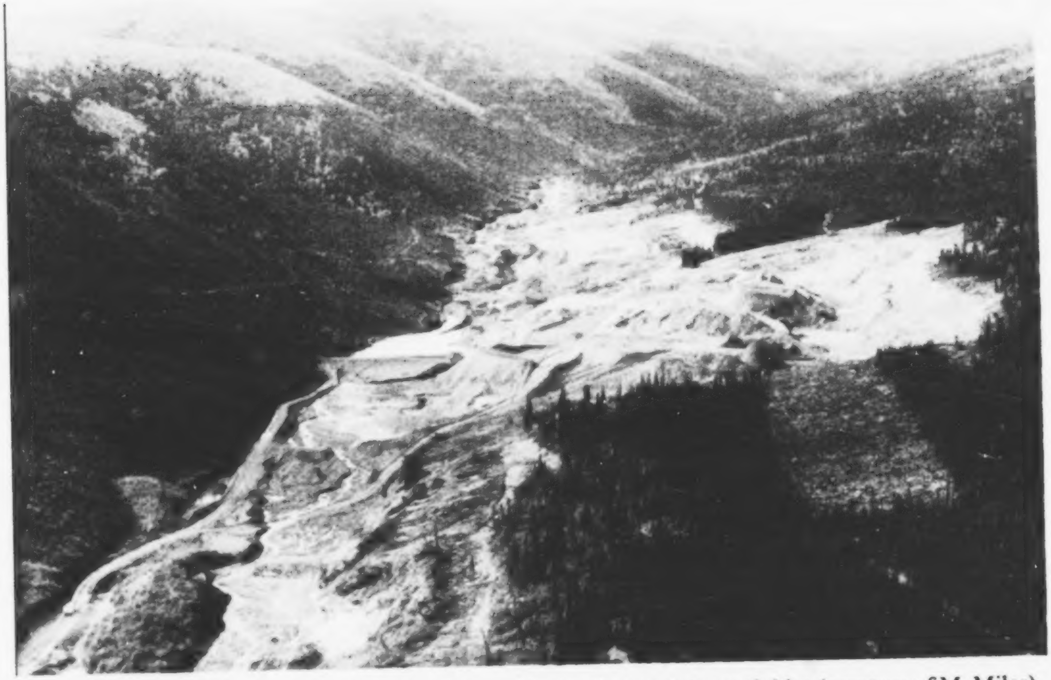


Figure 12. Disturbance to fish habitat and landscape due to placer mining activities (courtesy of M. Miles).

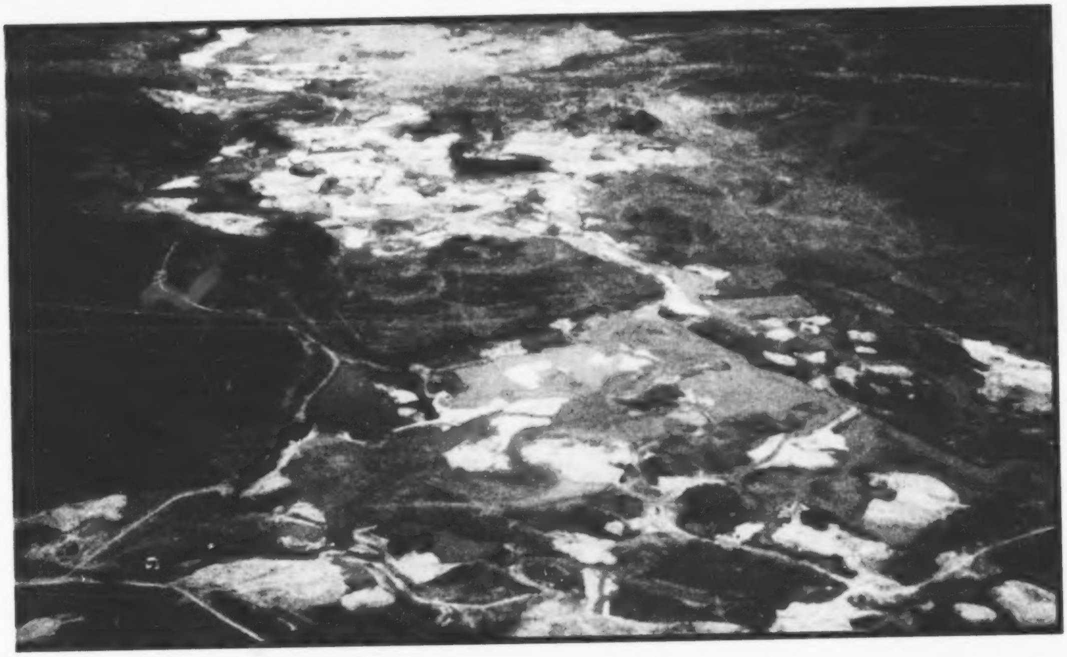


Figure 13. Extensive modification of the Indian River due to placer mining, Yukon Territory (courtesy of M. Miles).

It was estimated that in 1988 (Norecol Environmental Consultants Ltd. 1989) that 6-8 million m^3 of material were sluiced and 2-3 times that volume removed to expose the gold-bearing deposits, for a total of about 20-30 million m^3 of material moved. According to Norecol Environmental Consultants Ltd. (1989), the advances in technology and the abilities of miners to process large amounts of material because of machinery, has permitted extraction of gold from deposits in which concentrations were much lower (e.g. 0.02 oz/yd^3 or 0.015 oz/m^3) than those from the "break-even grade of 0.75 (raw oz)/yd^3 (equivalent to 0.57 oz/0.76 m^3) of pay gravel for hand-processed techniques that was necessary at the turn of the century" (i.e. early 1900s).

Environmental setting and scope: In 1990, there were 17,900 placer claims in "good standing," close to the 18,000 that existed at the peak of the Klondike gold rush in 1898 (Seakem Group Ltd. 1992). In 2003, a total of 324 valid water licences were in place to permit the mining for placer gold (Government of Yukon 2003). The expiry date of these licences ranged between 2004 and 2015, and approximately 6 operations had a zero sediment discharge stipulation, while the majority (over 85%) had a settleable solids (SS) criterion (see comments below regarding measurement of sediment in water and the appropriateness of this measure).

The extent to which the Yukon River drainage has been affected by placer mining has been estimated. In 1993, through the use of topographical maps, placer mining was estimated to have occurred in, and therefore affected, approximately 8% of watercourses of the Yukon River drainage (Al von Finster, Resource Restoration Biologist, and Steve Gotch, Senior Habitat Biologist, DFO, Whitehorse, YT; unpublished information). However, using more recent, refined digital mapping technology that incorporated length of watercourses affected (versus area) it was possible to estimate the total distance of all of the creeks, rivers and streams located downstream of valid placer claims which could be affected to some degree should mining activity occur (2003 data). The extent of affected waters in the 1993 assessment was based on the assumption that placer mining not only results in a direct disruption of habitat in a defined area, but also contributes sediments to all waters downstream of the site (irrespective of the quantification of effects). When the extent of the total distance of watercourses "affected" by mining activities was compared to the total distance of Yukon River watershed watercourses, the relative and approximate extent of affected waters was calculated to be 17%. This estimate represents 17% of the total linear distance of all watercourses within the Yukon River drainage that have been licensed for mining activity in the past or present, or that have had sediment released as a result of placer mining. Essentially, the 17% estimate indicates the distance of all watercourses within the Yukon River drainage which are located downstream of placer mining areas (Steve Gotch, Senior Habitat Biologist, DFO, Whitehorse, YT; unpublished information). The discrepancy between the 1993 estimate and the 2003 estimate is not due to a significant increase in mining activities, but rather, because the 1993 topographical assessment did not include the mainstem areas of major river systems within the Yukon River drainage (such as those of the Yukon or the White Rivers). A more detailed assessment of the total area disrupted through placer mining has not been undertaken to date due to the long history of activity within the territory (would require an extensive survey of historical records) as well as the vast areas that were likely disrupted at the beginning of the last century, but were likely poorly documented (Steve Gotch, Senior Habitat Biologist, DFO, Whitehorse, YT; pers. comm.).

The extent of mining may also be obtained from a review of licenses and an assessment of disruption to watercourses (Steve Gotch, Senior Habitat Biologist, DFO, Whitehorse, YT; unpublished information). Under the provisions of Yukon Placer Authorization (Government of Canada 1993) any streams designated as Type IV (streams designated as unimportant; refer to classification and designation of streams in text below) could be subjected to disturbance through mining activities, provided the operator procures a water use license from the Yukon Water Board and a Yukon mining land use authorization. Also, a number of streams (classified as Type I, II or III – declining levels of importance – see text below) are being subjected to disturbance through specific *Fisheries Act* s. 35(2) authorizations. There are approximately 487 Type IV streams that are currently designated under the provisions of the YPA.

As of 2003, there were 206 watercourses with valid water use licenses (some creeks have many licenses associated with them) and of these approximately 120-130 were considered active (i.e. actively clearing material, sluicing and/or producing). (Under the YPA a miner must have an "authorization" and a Yukon Territorial water use license in place prior to working within fish habitat and Yukon waters (mining land use permits are also required). The regulatory process involves: i) having the stream classified under the YPA; ii) procuring a water license which, in practice, is not issued unless a stream is classified; and iii) actively working the claim and producing. Because, typically, the licensing process takes several months, most mine operators hold their claims and licenses in valid standing, even though no mining activity is taking place. Those not mining full time may wait for the appropriate time to mine (Steve Gotch, Senior Habitat Biologist, DFO, Whitehorse, YT; pers. comm.).

Thus, depending upon the metrics used, the extent of disruption to fish habitat varies from kilometres of streams to percentages of watersheds. The net result is, however, a measurable amount of watercourses of the Yukon that are impacted by the mining activities.

Regulatory and governance issues associated with placer mining

Placer mining has been included in this report because of the need to consider the consistent application of the *Fisheries Act* in relation to industrial development (such as oil sands, placer and diamond mining). In addition to a focus for this assessment on consistency, we deal with aspects of compensation and restoration in a similar manner. A major technical issue is one of land movement and sediment generation, which is common to all three of these industries, and has been recognized as one of the most significant aquatic habitat issues worldwide (Waters 1995).

During the writing of this report in 2004 and 2005, changes were being negotiated and formulated regarding the regulation and administrative framework for placer mining. Brief mention will be made of these changes as they relate to the potential impacts of these mining activities on fish and their habitat in the Yukon, and the need for information that will assist the management processes. However, because this new regime will not be in place until at least 2007, the comments in this section are current (2005), and provide a status report on habitat management for this industrial sector.

Under the YPA, placer gold mining effectively operated under a "class authorization" thus distinguishing it from the legislative requirements imposed on the oil sands and diamond mining industries, and that part of the mining sector that falls under the Metal Mining Effluent Regulations (MMER) (Government of Canada 2002), and others who require specific authorizations to impact fish habitat.

The historical issues surrounding the administrative processes concerning placer gold mining have been the subjects of reviews since the initial regulatory document was signed in 1988. The document was based on a public consultative process and resulted in the Yukon Fisheries Protection Authorization (YFPA), an authorization under section 35 of the *Fisheries Act* (Government of Canada 1988). For administrative purposes, government and industry representatives constituted a committee (the Yukon Placer Mining Implementation Review Committee (IRC)) to oversee the application of the YFPA. The IRC reported to the respective Ministers of the Federal Departments of Indian and Northern Affairs Canada and of DFO.

The YFPA was reviewed in the early 1990s after a detailed study and evaluation of placer mining had been conducted under the auspices of the IRC. The review included an assessment of new knowledge and attempted to better understand and manage the placer industry with emphasis on fish and fish habitat. Details of the issues underpinning the management decisions together with more recent information are presented below.

The review of the YFPA resulted in a new regulatory framework document in 1993 named the Yukon Placer Authorization (YPA) (Government of Canada 1993). This document provided the processes and mechanisms for the management of placer mining and recognized the need for protection of fish and fish habitat. The YPA outlined, for example, methods for sediment measurement, the classification of streams to permit different levels of sediment discharges to watercourses, addressed fish habitat restoration and compensation, deferral of fishery values, diversions, etc., for mining to proceed. It was intended to permit placer mining to operate while providing protection to fish and habitat. This document was itself the subject of another review (that included the effectiveness and economic implications of the YPA) to develop recommendations for the Minister of DFO on proposed deferment of the water quality and fish habitat values. That report was completed in June 2002. In December 2002 the federal Minister of Fisheries and Oceans did not accept the recommendations (ISC 2004). As a consequence a new regime was sought that would address all concerns and provide a new direction for the management of placer mining and potential effects on fish and their habitat.

"In May 2003 the Council of Yukon First Nations (CYFN), Yukon Territorial

Government (YTG) and DFO signed a "Record of Agreement" (ROA) to work together and strive to reach consensus on a new regime for a sustainable placer industry that takes into account the importance of conservation and protection of fish and fish habitat. The ROA called for a "plan" by April 2004. The new regime is to be implemented on or before 2007" (ISC 2004), hence, in the interim period, placer mining operations are currently in accordance with the YPA (Government of Canada 1993).

The process of deciding upon a new regime is the focus of two committees – a Technical Working Committee that reports to a Steering Committee with members of the prime signatories represented in each committee. It was expected that the Implementation Steering Committee would develop a new "regime" that was based upon consideration of policy, legislative and technical/scientific issues (ISC 2004).

Within this new regime there is to be a recognition of the need to improve protection of fish habitat for "Yukon fisheries" (ISC 2004) and that "compensation and restoration are the primary means through which the policy goals (*Fisheries Act*) will be achieved where there are short-term disruptions of habitat" (ISC 2004). It was stated that the new regime would include "approaches to assessing habitat "health"; the relationship between habitat "health" and fish populations; identification of critical, important and marginal habitat; impacts of sediments on fish and fish habitats; and sediment measuring techniques."

The following addresses, in part, some of the scientific findings and issues related to the above comments and the requirements for appropriate knowledge that will assist in the management of fish and fish habitat in Yukon.

Habitat management considerations

The decision to permit placer gold mining to occur in and around watercourses is dependent upon many factors, including historical mining activities, "fisheries" values and socio-economic needs. To facilitate decision making it was deemed appropriate to assess the importance of watercourses. With respect to fish, objective and subjective assessments of their value by species and their use, and their presence or absence have been used to classify streams. The stream classification is then used to guide the application of sediment discharge criteria (or not) in the expectation that the habitat of these species will or will not be affected by sediments discharged. Implicit in this approach is the use of current knowledge and the application of assumptions regarding the impact of sediment on fish, the potential impact of activities, mitigation and compensatory measures, and the recovery (including restoration) of the productive capacity of habitats upon cessation of mining, etc. Thus activities are primarily regulated in relation to sediment discharge, and to a large degree have not previously included other aspects of habitat change such as stream integrity and destabilization, and compensation for the loss of the productive capacity of fish habitat.

With respect to the question as to whether protection of fish and their habitat will occur through an authorization, the answer lies not only in the appropriate setting and use of sediment criteria but also in the measure of compliance, and the use of best management practices. In addition, the value of species assemblages and the maintenance of the integrity of watersheds, and their recovery from mining activities demand attention. The concept of the deferment of the biological integrity of streams so that placer mining may proceed, and that such streams will recover their function following a cessation of mining is not well understood, documented, or researched. Thus the issue revolves around the relative importance of fish and their habitat and the needs of industry.

Issues that have been identified include those referenced above, the need to operate within the evolving governance structure in the Yukon and the development of an adaptive management approach that assures the ongoing refinement, and improvement of the process based upon experience and research.

Measurement and physical factors:

Settleable and suspended solids: Seakem Group Ltd. (1992) stated that "settleable solids" standards for this industry were inappropriate because the corresponding suspended sediment levels exceeded the thresholds for adverse effects on fish and their food resources. This statement is still valid, and reinforced by the studies of Miles and Associates (1993) who examined the relationships among settleable solids, suspended solids, and turbidity for 6 systems in the Yukon carrying settleable solids levels of 5 mL/L or less. The best relationship was between turbidity and suspended solids, followed by suspended sediment and settleable solids. The settleable solids/turbidity relationship was the poorest. The problem of using settleable solids determinations is exemplified through these data which reveal a poor correlation and even for settleable solids values <0.5 mL/L, the corresponding suspended sediment values ranged upwards of 2400 mg/L.

Because of the slow settling time for fine silts and clays and some organic particles, the volumetric determination of "settleable solids" may not accurately reflect the amount of sediment that a water sample may contain. Also, the practical detection limit (0.2 mL/L) is high, and most natural stream waters have less than this value for most of the year (Seakem Group Ltd. 1992). Furthermore, strong relationships between "settleable solids" and suspended sediment tend to be difficult to obtain, thus making this determination a much less useful measure of solids in a sample, and accordingly, of lower value for assessing effects on biota than the use of suspended sediment and turbidity upon which much research has been carried out. Despite these concerns Indian and Northern Affairs Canada (INAC) (1998) through Okanagan University College has been deriving relationships between settleable solids and suspended solids for particular waters in the Yukon. However, Norecol Environmental Consultants Ltd. (1989) reviewed the sediment effects literature and found no information on biological effects that employed this measure. It is clearly too variable a measure to quantify and correlate with suspended sediment and turbidity which have been examined in relation to their effects on fish.

Effluent concentrations from placer mines are unpredictable because of the large number of variables that affect the grain size and volume of discharged sediment. Well-designed settling ponds may initially discharge only clay and fine silt, but performance deteriorates as the deposited sediments reduce the pond volume. Sand as well as eroded and re-suspended finer sediments are discharged from full ponds.

Average daily sediment concentrations in placer-mined streams may include a considerable range of instantaneous concentrations, particularly close to the mines. For example, suspended sediment concentrations in placer-mined Clear Creek ranged from 300-1,800 mg/L in late May, and from 150-3,000 mg/L in late August (Seakem Group Ltd. 1992). In un-mined Moose Creek, suspended sediment was primarily moved during freshet when discharges were "high" and concentrations reached 100 mg/L. During summer low flows, suspended sediment concentrations were <10 mg/L and almost all of organic nature. Concentrations increased during rainstorm events to levels reached during freshet, but the total moved was much smaller.

Sediment loads were increased 40 or more times during freshet and 100 times or more during the sluicing season relative to an un-mined stream (Seakem Group Ltd. 1992). Results of sediment concentrations determined in streams in the Yukon revealed that values of settleable solids rarely were >0.5 mL/L in "pristine" water, but a maximum value of 120 mL/L at peak values (equivalent to 45,000 mg/L suspended sediment) was recorded in a stream subject to placer mining (Indian and Northern Affairs Canada) (INAC 1998). While settleable solids values were typically much lower than this maximum recorded value, the range in these determinations in "pristine" and mining waters illustrates the potential for adverse effects on aquatic organisms in the latter.

In placer-mined Clear Creek, the clay load alone provided an average suspended sediment concentration of 285 mg/L, a turbidity of 200-300 NTU, and a settleable solids value of <0.2 mL/L (the practical limit of detection is 0.2 mL/L) over the sluicing season (Seakem Group Ltd. 1992). Clay cannot easily be controlled by settling so this load represents the minimum that may be achieved in Clear Creek without additional technology.

Rates of sediment deposition in placer-mined Clear Creek ranged from 1-10 times higher than in Moose Creek, with maximum differences occurring during minimum flows (Seakem Group Ltd. 1992). In July, when flows were low and consistent on both creeks, the material deposited in traps was derived mostly from the suspended load and the dominant grain sizes were in the silt and clay particle size range. The overall deposition rate in Clear Creek was roughly 10 times that in Moose Creek, although this varied with grain size. The greatest increases occurred for medium and coarse sands, silts and clays.

Turbidity and settleable solids: Criteria that rely solely on the use of turbidity to protect aquatic organisms from elevated levels of suspended solids (United States Environmental Protection Agency (USEPA) 1986; Lloyd 1987) are not generally recommended for use because the typically site-specific and highly-significant relationship that exists between turbidity (NTUs) and suspended solids is not universally applicable. Turbidity determinations integrate the effects of suspended and dissolved material on the penetration of light in waters which, in turn, affects biological productivity (Lloyd et al. 1987). Depending upon the nature of the sediment in suspension and the dissolved material, the relationship between turbidity and suspended sediment will vary, usually by a factor less than 10. If, however, the relationship between suspended sediment concentrations and turbidity is known for a particular area (watershed, stream, reach, etc.), then turbidity *per se* may be used as a surrogate for suspended sediment and the appropriate criteria for the latter applied. Turbidity is, therefore, a useful, but approximate, indicator for suspended sediment.

Classification of streams and sediment discharges: The level of fish habitat protection (effluent guidelines and no net loss of habitat productive capacity) has been linked to stream classifications. Classifications have been based on the presence of fish species, their use of habitats, and the contribution the fish makes to an existing Yukon fishery.

Type I streams support spawning and incubation of salmon, rainbow trout and lake trout eggs and alevins. Proposed habitat disruptions must be compensated for before mining commences. Suspended sediment concentrations within effluent discharged from a placer mining operation into Type I watercourses may not exceed 0 mg/L above background concentrations. Suspended solid materials originating from upstream effluent sources must not exceed 25 mg/L within the Type I classified watercourse. In one circumstance a discharge of 400 mg/L suspended sediment into a Type I stream has been authorized under the YPA.

Type II streams support rearing salmon, rainbow trout and lake trout, for all or part of the year. Habitat disruptions must be compensated for prior to mining. The "end-of-pipe" discharge suspended sediment concentration for effluents deposited into Type II streams must be less than 200 mg/L of suspended solids. The receiving water quality objective, which is based on the contribution of suspended sediment from upstream sources, is not to exceed 25 mg/L above background levels. Deferment (see below) of Type II streams has been permitted resulting in effluent discharges of up to 5 mL/L of settleable solids into Type II classified watercourses. Because of the mL/L measurement (a volumetric determination with a poor correlation to suspended sediment – see below) and the nature of the materials discharged, this could equate to the release of many thousands of mg/L suspended sediment.

Type III streams support spawning, juvenile rearing and adults for those freshwater species that contribute to an existing Yukon fishery. Proposed habitat disruption must be compensated for prior to mining. Effluent discharge standards into these streams have been required to be <200mg/L with the receiving water quality objective of <25 mg/L. Deferment of water quality (see below) may occur which has allowed for effluent discharges of up to 5 mL/L directly into Type III streams.

Type IV streams have been designated as being unimportant and "not-essential to maintain existing fish populations that occupy the basins and habitat that will be affected." The YPA states "streams with no fish or streams with fish having no significant use by first nations, commercial, sport or domestic fisheries or not contributing to biological diversity" (i.e. for the maintenance of the current fisheries of the Yukon). Effluent discharge standards of up to 5 mL/L have been permitted, and receiving water quality objectives (those based on the upstream contribution of suspended sediment into the watercourse) are not considered. (Considering recent research findings and court decisions this is a debatable issue regarding the protection of such watercourses as indicated in the text below). Such streams can contain fish (e.g. Arctic grayling) deemed to be important in other regions. However, there are some qualifications. If the watercourse supports fish that do not currently contribute to a fishery or they have no significant use it will be designated as Type IVa and habitat restoration must occur when mining is complete. If fish are absent it will be classified as Type IVb and the stream channel must be stabilized when mining has been completed (fish habitat restoration is not required in Type IVb streams) and habitat restoration must occur upon completion of mining.

Type V streams are currently unclassified and accordingly they are treated as if they were classified as Type I.

Deferment of habitat: The deferment of fish habitat values on streams (where effluent standards are less restrictive than prescribed within the YPA) has been employed as a means to permit placer mining in locations where site conditions may preclude the establishment of adequate settling facilities. Deferrals have been categorized as Di and Dii, applicable to water quality, and water quality and habitat, respectively (note it is the opinion of the authors that water is an integral part of fish habitat).

A review of license information from the 2003 placer gold mining season identified that allowances for deviation from prescribed water quality standards within the YPA (general standards) occurred in 63% of Type III streams (effluent total suspended sediment (TSS) concentrations were >200 mg/L), 24% of Type II streams (effluent TSS concentrations >200 mg/L), and 33% of Type I streams (effluent TSS concentrations over 0 mg/L) for active mining operations. In some cases this occurred as a result of formal "Di" deferrals, where, as in other cases, this was a result of standards "grandfathered" within the YPA. Type IV streams are not considered to have "water quality deferrals" as operations within these areas are permitted to discharge up to the maximum allowable TSS concentration in mine effluents (5.0 mL/L).

In 2004, for streams classified as Type I, II, or III, there were 47 streams where TSS concentrations in mine effluents were permitted to exceed the generally-specified requirements. Without compensation and restoration, the productivity of these streams is expected to recover progressively through natural succession events once mining is completed.

The intent of this provision was for short-term use but the application of deferrals has occurred such that approximately 40% of current (2003) water use licenses for streams classified as Type I, II, or III, have deferred TSS water quality values (DFO; unpublished information). Deferments in downstream reaches of larger watercourses also permit a relaxation of water quality and effluent standards further upstream through the application of a placer effluent dilution model. This is used to determine effluent discharge standards for individual operations through a series of back calculations based on the dilution capacity of the system and the point of protection (downstream). Furthermore, there is no requirement for habitat compensation or restoration for the deferral of downstream water quality, and a water quality objective is not applied. Accordingly, effluent standards of 5 mL/L (thousands of mg/L; likely harmful to aquatic organisms (Newcombe and Jensen 1996; Birtwell 1999)) continue to be stipulated under the current management regime. To 2004, there had not been a deferral (Dii) of physical habitat on Type I, II, or III streams.

Assumptions that have contributed to the classification regime have included such factors as the dilution of sediment from small streams discharging into larger streams, the need to provide spawning grounds a higher level of protection, that salmonids are more sensitive or deserving (for socio-economic reasons) of greater protection, and that fish that currently support a fishery should be given greater protection than those that do not.

These assumptions present barriers to the decision-making process for the protection of fish and fish habitat as recent scientific evidence indicates that specific habitats such as the mouths of small streams, which include non-natal streams important for rearing salmonids, require a very high level of protection from suspended sediments. Furthermore, for the fish habitat provisions of the *Fisheries Act* to apply, fish do not have to be of a commercial or sporting value (Regina vs. Brown 23 CELR (N.S.) 227), and even relatively low suspended sediment levels have been shown to be harmful (e.g. Waters 1995; Anderson et al. 1996; Birtwell 1999).

Assumptions and related issues: Because of the limited extent of the information base that is used to manage fish and their habitat in relation to the effects of placer mining, a number of scientific issues require clarification and understanding. The following exemplify some of those considerations that require attention for the mutual benefit of industries and governments.

Reliance on the chinook salmon as a basis by which to monitor the status of habitat in relation to impacts of placer mining incorporates controlling variables that are impossible to account for. The majority of the life of these fish occurs in the marine environment and therefore outside of the control of factors in the Yukon. This comment is not intended to imply that chinook salmon are unimportant, but to indicate that many factors affect the survival of these fish and, accordingly, it is not possible to specifically relate, for example, the harvest of populations of adult salmonids to placer mining impacts. Impacts are most likely to be determined on the juveniles.

It has been assumed by some that there is "surplus" habitat in the Yukon and fish populations will not be affected by habitat changes due to mining. But, according to M.J. Bradford (Fisheries and Oceans, Canada, Co-operative Resource Management Institute, School of Resource and Environmental Management, Simon Fraser University, BC; pers.comm.) "available evidence does not support the notion of surplus habitat. Chinook salmon utilization of available streams under current escapement levels is high, suggesting that habitats are fully utilized. This assertion is supported by stock-recruit analysis, that shows current escapement targets are biologically based."

The relative lack of emphasis placed on the protection of Arctic grayling in placer mining areas cannot be supported based on research findings with respect to their sensitivity to sediment, for they have been determined to be another sediment-sensitive salmonid (see McLeay et al. 1987). Furthermore, this species has fidelity to feeding, spawning, and migration routes (Buzby and Deegan 2000), and loss of such habitats that support these life-cycle requirements would be expected to be significant to the survival of the particular stock. While one may speculate that this would also be so for some other freshwater species of fish, the information base upon which to draw such conclusions is inadequate for the task.

There is also the assumption that watercourses in which the habitat values have been deferred in favor of mining, will naturally recover from the effects of discharges of sediment, however, there is no information (studies) on the recovery of specific streams that have previously been "deferred." It is expected that recovery would occur, but the time frame for recovery would be difficult to predict and likely span tens or hundreds of years depending upon site-specific circumstances (see below).

Type IV streams are not subject to water quality standards, and can receive mine effluents with very high suspended sediment levels (3,000-15,000 mg/L). In general Type IV streams are headwaters of larger watercourses and it has been generally assumed that the discharge of high levels of sediments into the headwater streams will not impact fish and their habitat further downstream. The premise being related to sediment transport out of the system and the expected downstream "dilution" of these waters. However, Bradford (Fisheries and Oceans, Canada, Co-operative Resource Management Institute, School of Resource and Environmental Management, Simon Fraser University, BC; pers. comm.) stated that effluent discharges into Type IV streams have been shown to reduce, and in some cases eliminate primary and secondary productivity. These upper-watershed, headwater streams, while not always being occupied by fish, contribute to the life of organisms in downstream locations (such as streams classified as Types I, II, III). These watercourses supply water which is an indispensable component of the habitat of fish, but that aside, they provide the basic requirement of food (as covered under the habitat provisions of the *Fisheries Act*); a requirement that is susceptible to the discharge of sediment (Waters 1995; Birtwell 1999). Furthermore, Bradford's research has shown that juvenile fish use the confluence of Type IV streams and large rivers. Hence to accommodate such findings and concerns, sediment discharges require regulation, and need to be reduced from the now-permissible 5 mL/L (equivalent to many thousands of mg/L of suspended solids; concentrations that are known to adversely affect fish and their habitat).

Compensation and restoration: There has been no evaluation of the recovery of streams with deferred water quality standards (1993-present) following placer mining. There is a strong concern that natural restoration would not succeed (see comments below, and comments from Quigley and Harper and others in Consultations, Samis et al. 2005).

In the northern environment restoration challenges would be greater than in more temperate climes due to, for example, climate and permafrost, and a short growing season that make the restoration of streamside vegetation to pre-disruption conditions very challenging (S. Gotch, Senior Habitat Biologist, DFO, Whitehorse, YT; pers. comm.). Studies on Yukon streams (2-15 year mining period) show a time-dependent relationship regarding recovery. Also Alaska Department of Fish and Game (1987) concluded that recovery from placer mining would be a protracted process because of the lack of retention of moisture in soils that have had the smaller-sized particles removed (as occurs through sluicing).

Placer operators conducting restoration and compensation in Type IV streams are relying on a guideline that primarily focuses on short-term stream channel stability and erosion control. It is expected that the restoration of placer-mined streams is carried out in accordance with guidelines generated for that purpose (Hardy BBT Limited (1991)). For Type IVa streams the habitat productivity is to be restored when mining is complete. In the case of fish-less Type IVb streams the channel must be stabilized when mining is complete. However, there have been no evaluations of the efficacy of the YPA requirements for the restoration of Type IV streams. Studies of mines abandoned under earlier regulatory regimes suggest that natural recovery is a very long process, and may not occur at all. Whether the current mining circumstances will result in the restoration of productive capacity of streams is unknown.

Habitat recovery from mining: Hardy and Associates Ltd. (1981) assessed the recovery of fish habitat for Arctic grayling in the Yukon in regions that had been mined. They concluded that those physical characteristics such as water velocity, substrate and channel characteristics, were significant factors in the recovery processes. Elevated sediment loads may continue for many decades following cessation of placer mining as this material is transported to their receiving streams. Settling pond failure (Figure 14) following cessation of mining operations, or erosion of coarse tailings or stockpiled overburden may also provide substantial quantities of sediment to streams even in the absence of other placer mining activities (Seakem Group Ltd. 1992).

Hardy and Associates Ltd. (1981) deduced that to restore the physical habitat of placer-mined wide valleys to control levels would require from 29 to 72 years, and in narrow valleys no trends of habitat recovery were observed.

Water quality in wide valleys returned to control levels in approximately 20 years, whereas no predictable trend to recovery was found in narrow valleys.

Benthic community densities in wide and narrow valley streams recovered in 5 years following mining. However, the narrow valley streams did not return to control densities and became dominated with sediment-tolerant organisms. More recently, Bailey et al.

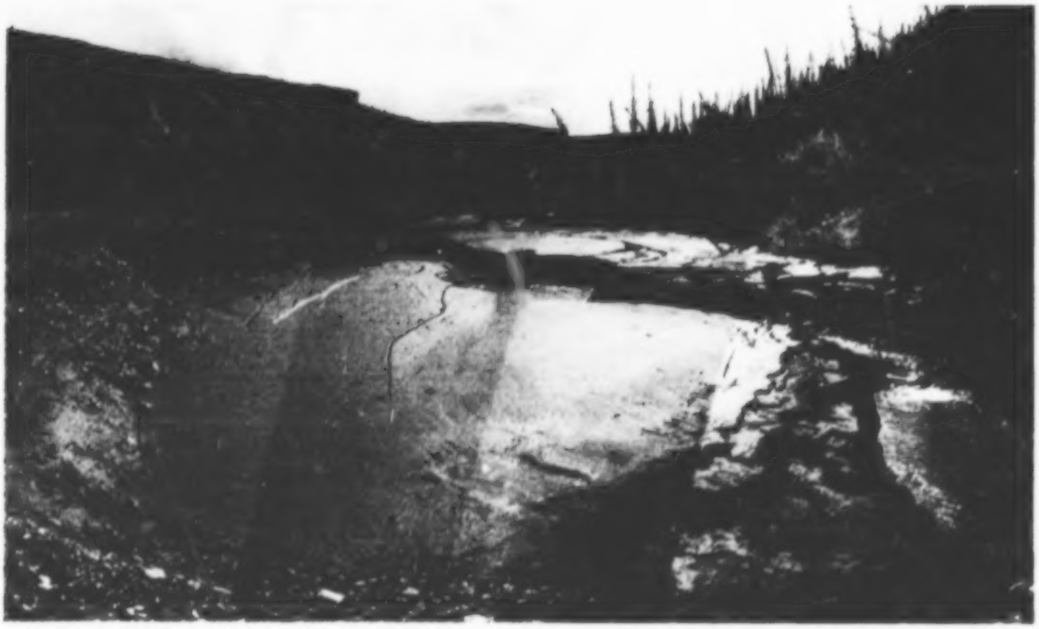


Figure 14. Landscape erosion and failure of placer mine settling pond (courtesy of M. Miles).

(1998) revealed impacts to stream invertebrates in watercourses that had been mined 10 years earlier.

The conclusion from this widespread assessment by Hardy and Associates Ltd. (1981) was that channel stability and placid flow areas were the most limiting factors associated with the recovery of fish habitat. This conclusion is similar to that deduced by the Alaska Department of Fish and Game (1987) which reported that lack of channel stability and elimination of habitat, including pools, undercut banks and backwater areas, were the greatest limitations to fish habitat recovery.

Scientific and technical considerations

Historical assessment of placer mining impacts in the Yukon: In the early 1990s, the IRC commissioned studies and reports on the status of placer mining and its impact upon fish and fish habitat within the Yukon. The conclusions from those reports (Seakem Group Ltd. 1992) provide a basis for the advancement of understanding of the effects of placer mining activities on fish and fish habitat.

Prior to the 1993 revisions to the YPA, numerous assessments were made in an attempt to justify changes to the previous authorization. It is the opinion of the writers that some of the conclusions reached during these assessments still have validity. As such they may be used in the formulation of information needs that will help guide future management decisions regarding the industry and the protection of fish and fish habitat. The findings of prior assessments also support those of the comprehensive review of placer mining that was undertaken by the Alaska Department of Fish and Game (1987) whose summary stated that there is strong evidence that placer gold mining can affect fish reproduction, growth, and survival. Further, streams subjected to placer mining may take many years to recover to their pre-disturbed condition.

Yukon Placer Mining Study (Seakem Group Ltd. 1992): Based upon the results of field studies the threshold for direct and indirect effects of sediments from placer mining on grayling and juvenile chinook salmon in natural environments was reported to be relatively low and <100 mg/L of suspended sediments. If this is so, the placer mining "regulations" are unlikely to be effective in protecting or enhancing fish habitats or fish populations in many regulated streams.

Benthic organisms (prey for fish): It was deduced that taxonomic diversity, density, and biomass of benthic macroinvertebrates (in a number of streams in the Yukon) were reduced in those waters subject to sediment from mining. Densities of benthic invertebrates were reduced by factors of up to 118 to 204 times at sites where mean concentrations of suspended sediment exceeded a threshold in the range of 50 mg/L to 175 mg/L relative to less-impacted streams.

Seakem Group Ltd. (1992) concluded that in areas with high suspended sediment concentrations, the suspended sediment threshold for major effects appeared to be in the range of 25 to 100 mg/L; the lowest concentration to have an observable effect was <75 mg/L. Reduced taxonomic diversity was also documented in the heavily-sedimented locations. These results support the findings of other researchers who have examined the effects of elevated levels of sediment from placer mining operations on aquatic organisms (Mathers et al. 1981; LaPerriere et al. 1983; Soroka and McKenzie-Grieve 1984; Wagener and LaPerriere 1985; Weber and Post 1985).

Fish: Investigations of fish populations and biomass provided evidence of the limiting effect that suspended sediments and associated turbidity may be having on the capacity of Arctic grayling and juvenile chinook salmon rearing habitat. Non-placer-mined streams (Moose, Stoney, and Flat Creeks) with turbidities ranging from 22-23 NTU supported a standing stock of fish that was 40 times greater than placer-affected Clear Creek (440-465 NTU; approximately equivalent to 500 mg/L suspended solids). The differences appeared to be attributable to variables affected by placer mining (turbidity/sediment load, presence/absence of pools and cover). Substrate composition, water depth and velocity, and stream size were generally comparable at all sites.

From the results of their field studies, Seakem Group Ltd. (1992) concluded the threshold for direct and indirect effects of sediment from placer mining on Arctic grayling and chinook salmon was <75 mg/L to 130 mg/L suspended solids.

Stream biological integrity: The effect of changes to stream integrity that were recorded by Seakem Group Ltd. (1992) for benthic invertebrates was also found in relation to fish. Overall, streams that had seasonal suspended sediment concentrations in excess of 50 mg/L did not support "significant" numbers of under-yearling Arctic grayling or chinook salmon.

Advances in, and application of, sediment effects research since 1990: Since the review of the Yukon Placer Authorization (YPA) in the early 1990s there has been an increase in the global literature that describes the responses of aquatic organisms to the addition of sediment in waters. For example, some research has focused on the effects of sediment on the avoidance by fish of turbid waters (Boubée et al. 1997), and on their feeding behavior and success (e.g. Vinyard and Yuan 1996; Abrahams and Kattenfeld 1997; Vogel and Beauchamp 1999). These and other new data serve to add to the already voluminous information on the topic.

A summary and management-related document that describes the effects of sediment on aquatic organisms was published by DFO (Birtwell 1999). This document also provides information on recently-derived criteria/guidelines designed to assist in the protection of aquatic organisms. Perhaps of increased significance in the recent guidelines is the recognition and documentation that increasing the duration of exposure of an organism to sediments, and turbid conditions, can result in increasing harm (Newcombe and Jensen 1996; Newcombe 2003).

In addition, new studies by Fisheries and Oceans Science staff (Ian Birtwell, Mike Bradford, and Jill Korstrom) have been carried out (Samis et al. 2005). Collectively, the information generated in recent years provides new and yet supportive information to that generated previously, and tends to refine our understanding of the effects of sediment on aquatic resources.

Duration of exposure to sediments and severity of effects: Since the early 1990s there has been more literature published that reveals the impacts of suspended sediment on fish, and most of this literature in recent years has been related to the impairment of their feeding through changes in water clarity at relatively low (tens to hundreds) levels of NTU).

Perhaps of most significance has been the derivation and acceptance (following initial criticism) of the concept that the effects of exposure to sediment are related to both the concentration and the duration of exposure. The concept was advanced in 1990 and refined in 1996. The longer the duration of exposure the greater the harm or "Severity of Ill Effect (SIE)." Thus, exposure to a high concentration for a short duration of time may produce an effect similar to that produced through exposure to lower sediment levels over a longer time period. This concept and its model has been tested and verified (e.g. Shaw and Richardson 2001; Birtwell et al. 2003; Korstrom and Birtwell 2005; Ian Birtwell and Jill Korstrom, DFO, West Vancouver Laboratory, West Vancouver, BC; unpublished data in Samis et al. 2005).

Development of guidelines and acceptable sediment levels: Because of the emergence of new information and the model development of Newcombe and Jensen (1996), new and updated water quality guidelines emerged in the late 1990s (British Columbia Ministry of Environment, Lands and Parks 1998; Canadian Council of Ministers of Environment, CCME 1999). These guidelines rely upon the current data at the time of their derivation and reveal the need for very stringent control of sediment (e.g. <5 mg/L suspended solids and 2 NTU turbidity over ambient) if one is to protect fish and their habitat. In recognition of the need to restrict the discharge of sediments in the 2002 MMER, under the *Fisheries Act*, established discharge limits of 15 mg/L total suspended solids (maximum monthly mean) and 30 mg/L total suspended solids (maximum within "grab" sample), which are more stringent than prior regulations.

Placer mining regulation has been guided by the deductions of the European Inland Fisheries Advisory Commission (1964), which provided comments on expected effects on fisheries within waters where certain suspended sediment concentrations occurred. The European Inland Fisheries Advisory Commission (1964) deduced that at <25 parts per million (ppm, which approximates mg/L of suspended solids) there was no evidence of harmful effects on fish and fisheries; at 25-80 ppm – it should be possible to maintain good to moderate fisheries, however the yield would be somewhat diminished relative to waters with <25 ppm suspended solids; at 80-400 ppm – these waters are unlikely to support good freshwater fisheries; and at >400 ppm suspended solids – at best, only poor fisheries are likely to be found. To guide decision making over the effects of placer mining sediments, it was recognized that there was some level of risk to aquatic organisms depending upon the sediment levels discharged and the sensitivity of the organisms in the receiving stream. After a review of available information it was determined that the impacts could be classified in relation to the levels of risk to which the fish habitat would be subjected, and that these impacts would be best assessed using increases in the concentration of suspended sediment above background levels. The levels of risk and the corresponding concentrations of sediment follow in Table 23.

Table 23. Levels of risk to fish and their habitat and the corresponding sediment concentrations (adapted from European Inland Fisheries Advisory Commission (1964)).

Sediment increase (mg/L)	Risk to fish and their habitat
0	No risk
<25	Very low risk
25-100	Low risk
100-200	Moderate risk
200-400	High risk
>400	Unacceptable risk

Recent guidelines adopted provincially and nationally (see Birtwell 1999) have utilized more recent information, and through assessments and data analysis it has been determined that there is the need for very stringent control of sediment (e.g. <5 mg/L suspended solids and 2 NTU turbidity for prolonged discharges) if one is to protect fish and their habitat.

Recent Fisheries and Oceans Science research related to placer mining: The following comments exemplify some of the research that has been necessary to assist with understanding fish ecology and the effects of sediment discharges upon fish and their habitat. Further work is required as detailed in other sections.

Science staff carried out research in the late 1990s to assist in the understanding of the ecology of species in northern Canada and to provide information on sediment effects. Two topics were considered, and the following comments outline some of that work to exemplify the type of focused yet basic research that has merit for enhanced understanding and assistance in decision making. The topics of the research were: the ecology of juvenile chinook salmon and their use of non-natal habitat (i.e. waters where they were not hatched), and an examination of the responses of juvenile chinook salmon to sediment. More detailed information on this research is provided in the appendix to this report (Samis et al. 2005).

Sediment effects on salmon: Exposure of chinook salmon to sediment impaired their escape fright response, and marginally affected their susceptibility to predation (Birtwell and Korstrom 2002*b*).

There were no demonstrable effects on the growth, feeding, biochemistry, and behavior of juvenile coho salmon exposed to mean sediment levels that were <5 mg/L, over a 6.5-month period (Birtwell et al. 2003). But there was an increased impairment of feeding behavior in chinook salmon with increased sediment concentration (>0-1,000 mg/L) and there was greater impairment with increased duration of exposure (determined at 3 and 9 weeks). Chinook salmon feeding efficiency on surface prey was impaired compared to that of control fish in the 3 sediment concentrations tested.

The growth of juvenile chinook salmon fed a commercial sinking ration was impaired by exposure to sediment, but the fish still grew. The behavior of chinook juveniles to gather food from the bottom, mid water, and surface of their habitat (in contrast to the typical feeding behavior of some other fish such as Arctic grayling), ensured that, over sufficient time, they could gather enough food for growth, even in the highest sediment concentration used (1,000 mg/L). However, in nature the opportunity to feed is quite different than that in the laboratory, and elevated sediment levels reduce the bottom-dwelling food organisms that are typically preyed upon by juvenile salmonids. Impoverished food resources typically occur in sediment-laden streams so it is likely that there will not be an abundance of food in such systems, and that feeding opportunities on bottom-dwelling organisms will be significantly reduced (the literature contains much information of this topic). Feeding by sight on organisms in the water column and on the water surface is significantly impaired at low sediment concentrations. It was deduced therefore, that juvenile chinook salmon would be able to feed and grow in waters with sediment concentrations as high as 1,000 mg/L as long as they were given sufficient time and food resources. Unfortunately both of these are very improbable in the wild.

Because elevated levels of suspended sediment have been shown to adversely impact prey items for these fish it is highly probable that the diminished feeding efficiency of fish in turbid waters will, when combined with diminishing food resources have a greater negative effect on growth, and hence survival, relative to fish in waters not so affected. These effects would be expected to be manifest at sediment concentrations <100 mg/L. Of practical concern here is the reduction in the productive capacity of headwater streams that may not have high fishery values or contain fish. These streams could be, and sometimes are, subjected to less-stringent sediment discharge standards, and accordingly impacts in upstream waters will likely impact waters downstream and organisms within them. Thus, the degradation of upstream watercourses will not necessarily safeguard the fish and fish habitat in more downstream reaches because sediment moves downstream in response to energy within the system.

Research on egg fertilization-suspended sediment interactions was conducted by testing the effect of suspended sediment on salmonid fertilization success (R. Galbraith, DFO, Co-operative Resource Management Institute, School of Resource and Environmental Management, Simon Fraser University, BC; unpublished data). Preliminary results suggest that suspended sediment concentrations greater than 2,000 mg/L reduce fertilization success in gametes of sockeye salmon (*Oncorhynchus nerka*), rainbow trout (*O. mykiss*), and coho salmon (*O. kisutch*).

Other research that has been carried out in recent years has tended to focus on the feeding of fishes and not specifically on their tolerance to sediment and survival. Sediment levels that are harmful to fish and their habitat are typically variable and relate to the species and life history stage. Low levels (10s of mg/L) can harm fish. The significance of this harm is a topic for debate because of the subjective understanding of significance.

Our previous research had shown that the Arctic grayling was a sensitive salmonid and studies in Alaska revealed that sac fry could be killed at concentrations of 22.5 mg/L over 48 h (LaPerriere and Reynolds 1997; LaPerriere, cited in Newcombe and Jensen 1996). On the other hand, juveniles survived exposure to thousands of mg/L of fine suspended sediment, yet evoked stress responses in this life cycle stage at tens of mg/L (McLeay et al. 1987).

One of the significant issues facing the regulation of sediment is that of the quantity of material discharged and its settlement in watercourses. Sediment does not degrade, as does organic material. Fine material will settle and remove/obliterate fish habitat (physical effects and food resources). In regard to this, Suttle et al. (2004) stated that "although excessive loading of fine sediments into rivers is well known to degrade salmonid spawning habitat, its effects on rearing juveniles have been unclear." They examined the responses of juvenile salmonids and the food webs supporting them by manipulating fine bed sediment in a river. Increasing concentrations of deposited fine sediment decreased growth and survival of juvenile steelhead trout. These findings were associated with a corresponding shift in invertebrates toward burrowing taxa that were unavailable as prey, and with increased steelhead activity and injury at higher levels of fine sediment. "The linear relationship between deposited fine sediment and juvenile steelhead growth suggests that there is no threshold below which exacerbation of fine-sediment delivery and storage in gravel bedded rivers will be harmless, but also that any reduction could produce immediate benefits for salmonid restoration" (Suttle et al. 2004).

The experiments of Suttle et al. (2004) demonstrate that "fine-sediment deposition, even at low concentrations, can decrease growth and survival of juvenile salmonids." They found no threshold below which fine-sediment addition was harmless; thus they concluded "fine-sediment deposition in steelhead bearing rivers in this region will further impair this potentially population-limiting life stage, while land management practices that decrease fine-sediment loading or storage in channels may benefit salmonid populations."

Settling ponds often fail and therefore the material that has been kept out of streams then is discharged, thus negating the effects of containment. So containment practices themselves often pose a problem for the maintenance of stream integrity.

While the discharge of low concentrations of suspended sediment may seem innocuous, the accumulation of the material and its secondary effect on light transmission and reducing aquatic productivity are potential effects of significance.

Concluding comments on sediment effects: It is concluded that elevated levels of sediment (typically over background) may be harmful to fish (e.g. acutely lethal, or elicit sublethal responses that compromise their well being and jeopardize survival), and in addition, negatively impact on their habitat (Birtwell 1999).

Lethal levels of sediment, determined through laboratory experimentation over different durations of exposure, typically range from hundreds to hundreds of thousands of mg/L suspended sediment, whereas sublethal effects are typically manifest in the tens to hundreds of mg/L suspended sediment. Some species of aquatic organisms are more tolerant of suspended and deposited sediment than others (Lloyd et al. 1987; Newcombe and MacDonald 1991; Caux et al. 1997), and this variation must be recognized when assessing potential effects.

Although elevated levels of suspended sediment elicit adverse responses in individual aquatic organisms, it is difficult to extrapolate effects to the population or ecosystem levels. However, the biological productivity of turbid systems has been shown to be less than that of non-turbid systems (Lloyd et al. 1987). Anthropogenic activities, such as some placer mining operations, have resulted in lowered densities of aquatic organisms in watersheds through the elevation of suspended and deposited sediments (refer to Alaska Department of Fish and Game 1987; Lloyd et al. 1987; Seakem Group Ltd. 1992).

Criteria, guidelines and recommendations, though having been formulated by different agencies, all tend to be mutually supportive. At the same time they have application limitations, especially relating to the protection of aquatic organisms from the effects of sediment concentrations in the ≤ tens of mg/L. Application of the criteria must be done while recognizing potential impacts on aquatic organisms at both the lethal and the sublethal level. Particle size and nature of the sediment must be considered as well. Bioassay information that reveals the lethal effect of sediment over a short period of time (such as 96 h) provides only a coarse indication of the effects of elevated levels of sediment in the wild. Accordingly, and when available, the more appropriate criteria which incorporate sublethal and lethal effects knowledge should be used.

Criteria documented in the report by Birtwell (1999) that are based on suspended sediment levels are appropriate and endorsed for use. Recent guidelines have related elevated sediment levels to the natural hydrological regimes in streams and the associated variation in suspended sediment concentrations (Canadian Council of Resource and Environment Ministers, CCREM 1987; British Columbia Ministry of Environment, Lands and Parks, BCMELP 1998; Canadian Council of Ministers of the Environment, CCME 1999). In addition, the use of risk criteria in relation to the elevation of sediment concentrations above background (Government of Canada 1993) have merit and are supportive of, and based on, earlier-published criteria.

Models that use sediment concentration and duration of exposure to predict harm (Newcombe and MacDonald 1991; Newcombe and Jensen 1996; Caux et al. 1997; BCMELP 1998; CCME 1999) reveal significant trends in increasing harm to fish and other aquatic organisms with increasing duration of exposure. Such trend identification is of value in predicting the potential effects of sediment on aquatic organisms, but caution must be exercised when assessing the effects of low concentrations (≤ tens of mg/L) of suspended sediment over protracted periods of time. Furthermore, it is likely that there would be increased variation in the response among individuals and life stages of organisms to the effects of elevated, but lower and sublethal levels of sediments, relative to less variable responses at higher sediment levels; the latter being due to a greater severity of effect and less scope for adaptation, tolerance and resistance. Because of this, judicious application of these models is warranted when assessing the potential impacts of exposure to low levels of suspended sediment.

We concur with the conclusion of Seakem Group Ltd. (1992) regarding the inappropriateness of using Imhoff cone determinations of settleable solids (mL/L) to approximate values of, for example, suspended sediment; the latter have been documented as valid entities in examinations of the effects of sediment on fish, the former have not. The threshold for adverse impacts due to suspended sediments on fish and their prey organisms appears to be below the level of detection for the Imhoff cone currently being used in some instances to regulate sediment discharges from placer mines in the Yukon. The settleable solids values for allowable discharges to Type II (Di – deferred), III (Di), and IV streams in the Yukon, are too high to protect fish populations and, indeed, allow discharges of sediments at concentrations which will almost certainly result in the destruction of food resources and denial of habitat because of, for example, avoidance behaviors. Our current research (Birtwell and Korstrom, DFO; unpublished data) has provided new information on the levels of sediment (and the duration of exposure) that affect juvenile chinook. The results support the existing body of sediment effects literature, and indicate that suspended sediment levels for the protection of fish and the integrity of their habitat must be in the tens of mg/L.

To protect fish and fish habitat in the Yukon, new regulatory standards are needed based on either suspended sediment concentration or turbidity, or a combination thereof. Furthermore, the application of models used to predict sediment discharge and receiving water levels must take into account the impact of loading because sediment, unlike organic effluents, does not degrade rapidly. In addition, the models must recognize and be sensitive to the natural hydrological/sediment regime and the nature of stream mixing, to ensure the protection of fish and the integrity of their habitat. In this context is the fundamental requirement to stabilize and restore watercourses upon the cessation of mining or after a predetermined period of time.

Biology of juvenile chinook salmon in relation to placer mining: The absence or paucity of information on the ecology and biology of juvenile chinook salmon in the Yukon provided a limitation to understanding the habitat requirements of these fish and, accordingly, assumptions were included in decisions regarding the acceptability of habitat disruption for placer mining. Because of the misconception over the lack of significance of non-natal streams to fish, they were potentially subjected to elevated disturbance and sediment discharges.

M. J. Bradford (Fisheries and Oceans, Canada, Co-operative Resource Management Institute, School of Resource and Environmental Management, Simon Fraser University, BC) undertook studies to provide new knowledge on the ecology of juvenile chinook salmon. His research studies revealed that juvenile chinook did not rear in the Yukon River downstream of the significant input of sediment from the White River, and that the fish made extensive use of clear tributary non-natal streams to feed and grow. He also recorded over-winter use of small non-natal streams. The following brief comments regarding this work have been provided by Bradford. He stated that it has been recognized for a number of years that juvenile salmon make extensive use of small, non-natal tributaries of larger rivers for rearing habitat, at least during the summer months. To further increase our understanding of the use of small streams studies were conducted from 1998-2001 in the Whitehorse, Minto and Dawson areas. The most intensive component of the study was completed at Croucher Creek, near Whitehorse, and that work is documented in Bradford et al. (2001).

The following is unpublished data from work conducted in the Minto and Dawson areas. The analyses are preliminary.

Fish use of non-natal streams: Fish were present in each of the 11 streams sampled in the summertime, and their densities appeared to be qualitatively related to the type of habitat in each stream. Fish densities greater than 0.8 fish/m^2 were found in streams with a well-developed pool-riffle structure, while streams with lower densities were either very shallow or contained little habitat complexity. Woody debris, in the form of fallen trees and root wads was an important feature of these streams as it provided cover for young salmon, and facilitated the development of small pools that were their preferred habitats (Mossop and Bradford 2004). Trees large enough to cause pool formation were at least 70 years old.

A comparison of the use of small tributary streams and the mainstem of the Yukon River: Sampling occurred in the Minto and Dawson areas of the Yukon River mainstem (areas that contrast in sediment loads: in the Minto area the suspended sediment concentrations are <20 mg/L during July and August, whereas in the Dawson area the loads range from 200-500 mg/L). In all cases the samples were taken from locations outside of the influence of tributary stream inflows. Juvenile chinook salmon were relatively abundant through the summer months in the Minto area, but were not caught in the Dawson region. Similar results were obtained with electrofishing along the shorelines of the Yukon River.

Over wintering: The over-winter use of small non-natal streams of the Yukon drainage by underyearling juvenile chinook salmon is not well understood. Surveys were conducted in 1999-2001 in a variety of areas to document potential over wintering by young salmon. In some cases, captures were made in mid winter through ice, but in most instances sampling was conducted in May, prior to the emigration of yearling smolts. Over wintering juvenile chinook salmon were found in most streams in the southern portion of the Yukon, but were absent from streams in the Dawson area.

Even at the relatively-low adult escapements of recent years, juvenile chinook salmon appear to make extensive use of small non-natal tributary streams in summer months. Juvenile salmon were found in the lower reaches of every stream sampled, unless there was a blockage from a windfall or beaver dam. It seems probable that the upstream distribution of salmon will vary annually because of changes in blockages, stream flows and fish densities. Bradford et al. (2001) provided detailed information on habitat use, and comparisons between the densities of fish in Yukon streams and other chinook-producing areas.

Small streams appear to be essential fish habitat in the Dawson area during summer. Bradford did not catch juveniles in the Yukon River mainstem, except at the mouths of creeks and rivers. It appears that the high sediment loads in this part of the Yukon River do not result in conditions suitable for juveniles. It is unknown whether suspended sediment conditions in the mainstem are attenuated further downstream in Alaska such that juveniles use mainstem habitats in downstream areas. Some juveniles were captured in the Yukon River near Dawson in the fall months when suspended sediment levels were much lower – these fish may have been those that left the small non-natal streams in this area before freeze-up.

Juvenile chinook salmon were present in tributary streams in relatively-high densities and the Yukon River mainstem in the Minto area, suggesting that the habitats are extensively used there.

The results of the over-winter survey suggest that among small non-natal streams over wintering is limited to the more southerly parts of the Yukon Basin, and corresponds to the extent of glaciation. The results suggest that the combination of milder climate and stronger groundwater flows from underlying glacial deposits provide conditions suitable for fish during winter. In any area there is likely a relationship between stream size and over-wintering capability, as larger streams are more likely to have adequate base flows in winter. Details on the size, survival and migration timing of yearling salmon are provided in Bradford et al. (2001).

Stream confluences: The impacts of placer mining effluent on fish habitat at the confluence of streams was been identified as a topic for review (in the 2001/02 YPA review). The following comments describe some recent research on fish habitat at the confluence of small clear-water tributaries and larger, turbid rivers. The work was conducted in the Dawson region, and concerned small streams that drain directly into the Yukon River mainstem. These areas were all unaffected by recent placer mining.

The objectives of the study were to estimate the density of juvenile chinook salmon in confluence areas, and compare these densities to those along the margins of the Yukon River away from tributary stream confluences; to provide data on the abundance of food organisms in the confluence relative to other areas of the Yukon River; and to describe the general physical conditions in terms of water temperature and clarity.

Yukon River suspended sediment levels ranged from 200 to 500 mg/L during the study, with major inputs from the White River and other tributaries. Water temperature decreased from July to August from 15-12 °C. With the exception of rainstorm-induced flood events, suspended sediment levels in the tributary streams were <10 mg/L, and water temperature varied from 4-7 °C. The mixing zones of visibly-intermediate turbidities extended 10-20 metres downstream from each tributary, in a band 2-3 m in width.

The results of this survey suggest that the areas of confluence of small streams and the Yukon River are important fish habitat. Chinook salmon juveniles emerge from spawning gravels in their natal streams in mid May, and some move downstream. They appear to enter small creeks in early July, after having grown from 0.3 g to about 1.2 g during their 6-week downstream migration. These results and those for 1999 and 2000 (Bradford unpublished data), show that creek mouths are the only habitats of the Yukon River mainstem that are used by salmon in the Dawson region in July and August. In some cases, fish probably leave the confluence area and move upstream in the tributary stream. However, in the case of Montana Creek, which is too small and steep to support rearing salmon, the high densities of salmon that were found at the creek mouth suggest that fish may use the confluences of very small streams while in transit to other locales downstream.

The confluence areas likely offer better feeding conditions for young salmon than other areas of the Yukon River mainstem. The drift sampling data showed that more organisms were present in the water in the confluent waters than in other areas. However, the densities of organisms in the substrate of each area were not different, suggesting that the increase in drifting organisms results from input from the tributary stream to the confluence. It seems also likely that the lower turbidity in the confluence of clear water streams would offer better conditions for visual-based foraging than undiluted Yukon River water.

ASPECTS OF FISH HABITAT MANAGEMENT (INCLUDING MITIGATION, COMPENSATION AND RESTORATION)

This section of the report is concerned with some of the fundamental aspects of habitat management: those which are associated with developments that impact fish habitat and the related scientific challenges and requirements. Information is also provided on the use of streams and lakes for industrial activities and, with respect to the mining industry in Canada, a detailed accounting of lakes and streams that have been eliminated and degraded and the associated compensatory measures. Aspects of the restoration of fish habitat are also considered. Placer gold mining, which has been operating the longest of the mining activities in Canada, exemplifies some of the ubiquitous sediment and other issues and challenges for habitat management in northern Canada.

Aside from the ecological importance and challenges to compensate for, and restore degraded fish habitat, the importance of the application of the Habitat Policy in achieving no net loss of the productive capacity was emphasized through the Auditor General's Report to the House of Commons (Government of Canada 1997). Within this report it is stated that "Fisheries and Oceans should devote more time and effort to compliance monitoring and follow-up in order to assess the effects of its habitat management decisions and its performance toward the achievement of "no net loss" of habitat." Evaluations have been carried out within the last 5 years and the results have indicated a need to address many facets of these activities if the objectives of the Habitat Policy are to be met regarding the no net loss of fish habitat productive capacity.

In the prevailing cold northern environment it is expected that fish habitat compensatory and restorative measures would occur over protracted periods of time relative to those in more temperate locations. Accordingly it will be necessary to choose appropriate strategies and to evaluate them in a manner that is commensurate with this understanding.

It has been recognized that good habitat is a fundamental requirement in order to have a good fishery, and if biological (biotic) and non-biological (abiotic) requirements are at optimal levels then growth and survival of fish in the population would be maximized (Hayes et al. 1996). Furthermore, the habitat requirements of a fish population are directly related to the requirements of an individual's growth and survival, i.e. its niche. Accordingly, habitat requirements may vary over the life of an individual and can also vary between individuals of the same species (Hayes et al. 1996).

"Habitat is simply the place where an organism lives….Physical, chemical and biological variables (the environment) define the place where an organism lives. Niche, a closely related term defines the way a species adjusts to other related species in this space" (Hudson et al. 1992, cited in Hayes et al. 1996). It follows, therefore, that because biotic and abiotic factors play an important role in determining fish growth and survival, a definition of habitat that includes both components is essential to manage and understand the relationship between habitat quality and quantity and fish population dynamics (Hayes et al. 1996*)*. The definitions of fish and fish habitat within the *Fisheries Act* are germane to this position.

Within the *Fisheries Act* the definition of fish includes "parts of fish, and shellfish, crustaceans, marine animals and any parts of shellfish, crustaceans or marine animals, and the eggs, sperm, spawn, larvae, spat and juvenile stages of fish, shellfish, crustaceans and marine animals." Fish habitat means, "spawning grounds and nursery, rearing, food supply and migration areas on which fish depend directly or indirectly in order to carry out their life processes." Accordingly, the definition recognizes the importance of aquatic areas but does not explicitly stipulate water and its quality as being part of fish habitat, yet from the perspective of the basic needs of, and classification of, aquatic organisms, water is the primary and essential component of their habitat. It follows therefore that changes in the habitat of fish will include changes in water *per se*. Section 36 of the *Fisheries Act* prohibits the deposit of a deleterious substance into water frequented by fish (i.e. "any substance that, if added to any water, would degrade or alter or form part of a process of degradation or alteration of the quality of that water so that it is rendered or is likely to be rendered deleterious to fish or fish habitat or to the use by man of fish that frequent that water"). In this way the *Fisheries Act* addresses and protects the water component of fish habitat in its full context.

Similarly, other provisions (section 35(1)) within the *Fisheries Act* consider the function that habitat plays in the perpetuation of fish, and within this context "no person shall carry on any work or undertaking that results in the harmful alteration, disruption or destruction of fish habitat" (spawning grounds and nursery, rearing, food supply and migration areas on which fish depend directly or indirectly in order to carry out their life processes); thus the linkages between the environment and fish are recognized.

Logically, negative changes to fish habitat would have adverse consequences to aquatic organisms, and fish habitat management practices typically involve strategies to avoid such harm. Avoidance of impacts, followed by mitigation of residual adverse effects to fish habitat are the first considerations to be taken when habitat loss is likely to be encountered. In order to guide habitat practitioners who must make decisions regarding the management of fish habitat and ensure that there is compliance with the intent of the *Fisheries Act*, DFO prepared a policy document (Department of Fisheries and Oceans 1986). The guiding principle of the Habitat Policy is to ensure that there is "no net loss of productive capacity of fish habitats" (refer to Minns 1997*a*). Thus, a key analytical step of habitat management is to determine whether and how a habitat manipulation, either inadvertently or deliberately, will affect productive capacity (Jones et al. 1996).

Productive capacity is defined in the Habitat Policy (Department of Fisheries and Oceans 1986) as "the maximum natural capability of habitats to produce healthy fish, safe for human consumption, or to support or produce aquatic organisms upon which fish depend." As Minns (1997*b*) points out, "the productive capacity is a potential, vested in the habitat and independent of the extant stocks of fish or associated organisms."

Measurement and quantification of productive capacity is required in order to meet the NNL principle of the Habitat Policy which is to maintain the long-term productive capacity. Accordingly, when more fish are available or able to exploit habitat the productive capacity must be sufficient to support them (Fisheries and Oceans Canada 1998).

According to guidance documents (e.g. Fisheries and Oceans Canada 1998) productive capacity is the measure of the capability of a habitat to produce fish and/or food organisms in natural or restored conditions. Within this document productive capacity is considered to be analogous to carrying capacity, (defined as the maximum biomass of organisms that can be sustained on a long-term basis by a given habitat).

The determination of productive capacity inevitably involves more than an assessment of the number of fish utilizing a given habitat at one specific time or the current level of fisheries production from the habitat (Fisheries and Oceans Canada 1998). The concept of productive capacity encompasses cyclical variations in habitat utilization as well as cycles in fish production.

Typically, habitat lost through project development or created through compensation is assessed and described according to physical parameters (e.g. area or volume of habitat; littoral zone; mudflat; salt marsh; gravel bed) combined with a biological classification (e.g. spawning grounds and nursery, rearing, food supply and migration areas).

Fisheries and Oceans Canada (2004*a*) stated that physical and biological descriptors may have to be used as surrogate measures of productive capacity until ongoing research provides more precise "tools." The development and provision of such techniques and their verification have been considered over numerous years, and at the present time there still remains the need for such research (e.g. DFO 2004). These aspects are addressed below in relation to technical considerations.

Management of habitat impacts

In determining the severity of potential impacts of projects on fish and fish habitat, the factors that DFO takes into consideration include the potential for the project to affect fish, fish habitat and/or people's use of fish as well as the nature of the effect (e.g. physical disturbance, temperature change, flow alteration, release of nutrients or contaminants, reduction in dissolved oxygen, obstruction of fish migration, alteration of oceanographic or limnological regime); the presence or abundance of a fish species which is actively, or has the potential to be, harvested in a subsistence, commercial and/or recreational fishery; whether the species (or fish population) at risk is considered as vulnerable in the context of the proposed project; the productive capacity (or surrogate habitat value) of the habitat and/or the degree to which it supports an important lifecycle process (e.g. spawning grounds and nursery, rearing, food supply and migration areas); the availability and anticipated effectiveness of mitigation and habitat compensation; proportion of the habitat with a similar productive capacity and contribution to the fish stock which may be affected; the habitat's resilience to damage and the amount of time it would need to recover, and, all other factors staff deem important to consider on a project-specific basis (Fisheries and Oceans Canada 1998).

Fish management (exploited and unexploited resources) and habitat area plans

Fish management and fish habitat area plans are fundamental to the successful management of fishery resources and the attainment of the productivity and conservation goals identified within the Habitat Policy. They represent the starting point for sound management that is stated in the Habitat Policy.

Fish habitat area plans provide the means whereby the goals of the Habitat Policy may be realized. They provide the basis for conserving the current and future productivity of an area (Minns 1997*b*). It is also recognized in the Habitat Policy that "natural resource interests such as the forestry, fishing, mining, energy, and agricultural sectors make legitimate demands on water resources, and that ways must be found to reconcile differences of opinion on the best use of those resources. Effective integration of resource sector objectives, including fisheries, will therefore involve co-operation and consultation with other government agencies and natural resource users" (Department of Fisheries and Oceans 1986). Thus, there is an explicit understanding that integrated management will occur. There is also an understanding that this requires an ecosystem approach when one considers that fish productivity is dependent upon the productivity of the system as a whole (Bisson 1989; Minns 1997*b*; Northcote and Hartman 2004). However, rigorous scientific studies are necessary to provide the information that will permit such an understanding to occur.

Depending upon the management objectives for a particular region, larger or smaller frames of reference will need to be applied to understand the relative effects due to habitat change. But in each situation the change should be quantifiable to ensure that the most appropriate decision is made. The task is likely more difficult the larger and more complex the system affected.

Clearly, if loss of fish and aquatic habitat occurred, it is most likely that there would be a negative affect on aquatic production at some point in time. Such losses may be acceptable from an socio-economic perspective, but they would set a threshold for loss that is contrary to the primary objectives of the Habitat Policy, unless, of course, suitable and successful compensation and mitigation occurred. In order to address the latter one must first know what is being affected and how. If information is inadequate for this initial step, mitigation and compensatory efforts will inevitably be hindered and likely compromised. Therefore, the basic knowledge of species and assemblages, and their physical and functional habitat requirements is fundamental to an examination of potential effects of habitat change or its elimination; requirements that transcend tiered assessments culminating in population-level effects and ecosystem productivity.

Addressing effects at different levels of biological organization will permit a more holistic understanding in which the cumulative impacts of even small changes to habitat will be recognized. According to Hayes et al. (1996), predictions of changes to habitat can only be done by approaching habitat analysis from such an holistic viewpoint that includes not only the physical and chemical components of habitat but also the biological components. The reasoning being that an evaluation of one without the other will produce an erroneous result as it is the interaction between the two that creates the production potential for fish populations. Hayes et al. (1996) exemplify their statement in reference to the synergism between the suitability of food and temperature "resources" (temperature alters growth rate, and food abundance similarly). Because changes in thermal habitat can be offset by changes in food abundance, assessments of growth could be due to either component. Thus, it is important to assess significant environmental features and their potential synergistic or antagonistic effects to provide accurate assessments of their interactions on populations of fish. This conclusion is supported by the comments of Ryder and Kerr (1989, cited in Lewis et al. 1996) who state that unless there is baseline knowledge of the state of major environmental determinants, quantification of the effects of habitat alteration will be difficult. This comment is equally relevant to the understanding of the restoration of habitats (refer to Kelso 1996).

Seldom have managers looked for a net gain in terms of increased standing stock of desired species over time, as long as a local and (or) short-term benefit has been observed. One reason for this small-scale focus for habitat management may relate to our understanding of its link to populations (Lewis et al. 1996).

Compensation

Provision exists for the use of discretion when assessing the potential effects of human activities on fish habitat, but when mitigation measures will not negate all adverse effects, compensation and other measures are considered (Fisheries and Oceans Canada 2004*a*). However, compensation is not a feasible option when considering the degradation of water frequented by fish by, for example, liquid waste discharges, contaminants, etc. The restoration of chemically-contaminated sites may be recognized as compensation, but only in rare circumstances (Fisheries and Oceans Canada 2004*a*).

The Fisheries and Oceans Canada (2004*a*) document provides guidance regarding considerations for projects that will likely result in harmful alteration, disruption or destruction of fish habitat (HADD) and that will require compensation in order to achieve NNL of the productive capacity of fish habitat. Compensation is defined in the Habitat Policy as "The replacement of natural habitat, increase in the productivity of existing habitat, or maintenance of fish production by artificial means in circumstances dictated by social and economic conditions, where mitigation techniques and other measures are not adequate to maintain habitats for Canada's fisheries resources." (Although the principle of NNL is not routinely considered as a statutory requirement, habitat compensation is generally required when a *Fisheries Act* s. 35(2) authorization will be issued. The habitat compensation contained as part of that authorization is needed to meet the Habitat Policy guiding principle of NNL of the productive capacity of fish habitat). Compensation would be consistent with local fish management plans, and it is expected that "partner agency objectives should be considered." It was expected that this document would assist in the application of "fair and consistent decisions on compensation matters across Canada, while recognizing the need for local discretion" (Fisheries and Oceans Canada 2004*a*). Some of the other comments in the "Practitioners Guide" are summarized below.

The decision to authorize a HADD is at the discretion of the DFO Minister and Departmental staff. The requirement of compensation is policy based and is made after the acceptability of the HADD is determined. If a HADD has been authorized conditional upon the completion of compensation, a failure to complete that compensation could invalidate the authorization. This would, in effect, leave the proponent with a HADD that was not authorized and is, therefore, potentially subject to prosecution pursuant to section 35(1) of the *Fisheries Act*.

Habitat compensation options: Fisheries and Oceans Canada (2004*a*) stated that local fisheries management plans should be used to help determine appropriate compensatory options for HADDs. However, a hierarchy of options is provided to facilitate decisions on compensation. Cash, in lieu of compensation, is considered to be unacceptable because "habitat compensation does not include financial means for compensating for tangible economic losses but deals only with actions intended to maintain the net production potential of fish habitat."

Fisheries and Oceans Canada (2004*a*) states that scientific tools to determine NNL through compensation ratios (created:lost) should be used where they are available and suitable (e.g. the Minns et al. 2001 "Habitat Suitability Matrix"). Higher (weighted) ratios are justified when success is less certain, there is increased variance in the quality of the fish habitat being replaced, and there is recognition of the time for the created habitat to become "functional."

The hierarchy of choices presented in Fisheries and Oceans Canada (2004*a*) is i) create or increase the productive capacity of like-for-like habitat in the same ecological unit ("populations of organisms considered together with their physical environment and the interacting processes amongst them"). "Like-for-like" compensation should aim to create or enhance habitat that has the same natural integrity, structure, and function of the habitat that was adversely affected; ii) create or increase the productive capacity of unlike habitat in the same ecological unit, an approach in which "there is less assurance that a "No Net Loss" has been achieved"; iii) create or increase the productive capacity of habitat in a different ecological unit, which may compromise biodiversity objectives since the HADD will affect one distinct population while the compensation will provide benefits to different species; and iv) use artificial production techniques to maintain a stock of fish, deferred compensation or restoration of chemically-contaminated sites. The final compensation plan could include elements at more than one level, with some like-for-like habitat and some unlike habitat creation, and the "ecological value of the existing habitat must be considered" at each level.

Although the guidance document provides sequential decision-making options there is provision for the adoption of alternative measures. Such circumstances would include "situations where site-specific issues are well understood and…where limitations to productive capacity are known, and local management plans containing clear objectives for the fishery may have been prepared." Also, discretion may be used if opportunities exist for maximizing the amount of habitat gained, particularly where there are known limitations (or "bottlenecks").

Restoration

Compensation may also include the restoration of degraded habitats. Here, restoration has been defined as "the treatment or clean-up of fish habitat that has been altered, disrupted or degraded for the purpose of increasing its capability to sustain a productive fisheries resource" (Fisheries and Oceans Canada 2004*a*). But, this assumes that the degradation does not involve the ongoing, unregulated deposit of a deleterious substance (which is not a candidate for compensation under the Habitat Policy). Restoration can include remedial action during or following mining activities such as the return of open pits and placer-mined streams to productive fish habitat. Compensation may also include the restoration of degraded habitats and may be specified within a *Fisheries Act* authorization.

According to Bradshaw (1996), restoration applies to a return to an original state, and that it should be thought of as applying to whole ecosystems. That is, it includes water and its quality. Pastorek et al. (1997) provide a similar understanding and consider that restoration returns an ecosystem to a close approximation of its condition before it was disturbed (an issue not to be confused with rehabilitation which improves a system to a "good working order" (Pastorek et al. 1997)).

The successful restoration of fish habitat is as dependent on the requirement for knowledge as is the effective compensation for changes and losses to fish habitat. The successful restoration of habitat is closely linked to our understanding of ecological processes to the extent that if we do not understand the processes we are unlikely to be successful in efforts to make the restoration work (Bradshaw 1996). The criteria for successful restoration require such an understanding.

Monitoring and research

DFO requires that monitoring of habitat changes occurs to determine whether the conditions of compensation plans have been met, and how new habitat is functioning. Implicit in this requirement is the need for more detailed monitoring when dealing with more complex issues. Also, it is expected that DFO would provide details and assessment of monitoring requirements and findings. However, the proponent has the responsibility for habitat compensation and monitoring, but DFO and others may carry out additional measures that would result in a net gain of productive capacity. The duration of monitoring is at the discretion of DFO, and it is dependent upon the complexity or uncertainty of the undertakings and should continue long enough to determine whether the compensatory measures are "functioning properly" (Fisheries and Oceans Canada 2004*a*).

DFO is to conduct research to "assess the relative importance of specific habitats as a factor contributing to fish production; assess the effects of human-induced chemical, physical and biological changes on fisheries resources and the habitats that support them; determine how adverse effects on fish habitat may be mitigated and establish criteria for the continued natural production and safe consumption of fish; develop and refine techniques to restore degraded and develop new habitats; refine our understanding of the factors that control the productive capacity of natural habitats and how to measure those factors; and develop improved methods of evaluating the economic and social worth of fish habitats" (Department of Fisheries and Oceans 1986).

Because the complexity of such research programs has been recognized, co-operation with other federal government departments, provincial and territorial agencies, and industry groups and associations is encouraged. It is also expected that habitat-related research priorities will be established through consultative arrangements with fishery managers, habitat managers, and where appropriate, industrial interests, government agencies, and the general public (Department of Fisheries and Oceans 1986).

The consultations and comments obtained during the preparation of this report support that concept for co-operative research ventures.

Knowledge requirements and practical application

There is a fundamental requirement for knowledge of the function of fish habitat in order to effectively mitigate potential damage to fish habitat, to compensate for its degradation and loss, and to rehabilitate and restore degraded habitat. Without this basic understanding, successful mitigation, compensatory and restorative measures will be compromised.

The maintenance or enhancement of fish production requires knowledge of at least the factors controlling production and the relative influence of habitat variables most influencing the fish (Rabeni and Sowa 1996; Kelso et al. 2001). Furthermore, successful habitat conservation and restoration must be biologically tested (Rabeni and Sowa 1996). Habitat practitioners must be able to identify important habitat within, for example, a watershed and how and why changes of land use will modify habitat. Accordingly, cause-response information is extremely important for input into resource and land-use planning and ecosystem restoration (Imhof et al. 1996). Simply put, there is a need to understand how organisms live within their environment and how they interact with it.

Inevitably there will be a continuing need for knowledge, and the identification of uncertainties in decisions that affect fish habitat offers opportunities for adaptive management (Holling 1978; Walters 1986). Habitat management issues frequently offer such opportunities because similar habitat manipulations are repeated thereby affording the opportunity for experimental design and the replication of adaptive experiments (Jones et al. 1996). For instance, habitat change or destruction may be allowed for which the "best guess" suggested no likely detrimental effect on a factor(s) of interest, but for which there is a substantial hidden risk. Conversely, conservative decisions may be made because the highly uncertain "best guess" suggests a detrimental impact. Ignoring key uncertainties would, therefore, squander the opportunity to learn and thereby reduce uncertainties for future decisions that concern the same issues (Jones et al. 1996).

Jones et al. (1996) identified processes that could assist in habitat management decision making. The approach comprises four components, namely the determination of management objectives, the identification of indicators, the analysis of cause-effect pathways linking habitat changes to ecological effects, and the determination of strategies to effect desirable habitat change. The core of this process is the use of a network of cause-effect linkages leading from habitat change to ecological effects, to ensure rigorous assessment of possible effects.

It is only logical that the analysis of all habitat management issues should begin with a careful articulation of ecological objectives or targets (Jones et al. 1996). These authors also recognize that each habitat management issue will have specific objectives derived from the general goals for ecosystem management. These goals are to maintain or increase (optimize) the harvest or yield of fish from an ecosystem, to protect or conserve healthy ecosystems from the effects of human activities, to preserve pristine or undisturbed natural systems and to restore degraded ecosystems to a healthier state (Jones et al. 1996). Similarly, Imhof et al. (1996) provided a hierarchical evaluation system for determining the relations between processes that generate physical features and how these features are used by fish; information that they considered to be essential for long-term management of fish habitat within watersheds. The model proposed is hierarchical including watershed, reach, and site scales. Physical characteristics at these scales are separated into attributes, features, or variables that provide information on cause-response relationships. A functional analysis of physical characteristics and processes that generate physical habitat and a functional analysis of habitat requirements are related to the life cycle of an indicator species. Concepts of "health," "integrity," and "fit" are used to assess the physical states and conditions of the environment to determine the potential fit for a species based on its life-cycle requirements. Imhof et al. (1996) contended that standardized methodologies exist for fish habitat assessment (e.g. Armantrout 1981; Platts et al. 1983), whereas tools to assist in determining the impact of human disturbances, the scale of controlling impact, and their cause-response relationships are limited. While some standardized and defensible methodologies exist (e.g. Minns et al. 1995) there remains much-needed research validation and evaluation, especially in regards to the more remote regions of Canada now experiencing the effects of human developments.

Technical and scientific considerations

The advent of the Habitat Policy (Department of Fisheries and Oceans 1986) stimulated an increased need for research into the measurement of the productive capacity (PC) of fish habitat. It was considered to be the most important and most difficult aspect of habitat management (Levings et al. 1997*a*, 1997*b*). Minns et al. (1996) point out that habitat alteration may be expected to invoke 4 community or ecosystem responses. That is, total fish biomass and production changes, alteration of fish assemblage composition (species richness, distribution of biomass, production by species), and the alteration of the distribution of all or some elements of the assemblage in time, and (or) space, and a response in the non-fish biotic elements of the "target" ecosystem. However, variability of structure and function in space and time will act to confound the detection of ecosystem change (Minns et al. 1996). In this regard, even though the Habitat Policy requires measures of PC, Minns (1997*b*) considered that direct measures of productive capacity will always be a "luxury," and that reliance should be placed upon "mapable" habitat indicators, and research and audits should be relied upon for validation. Minns (1997*b*) also provided comments on the need to quantify the productivity of fish habitat in recognition of its role in the productivity of fish. He stated that little progress has been made to sustain productivity, and that "most extinctions of fish have been due to habitat destruction." Thus the challenge exists not only for habitat to be maintained and improved to benefit fish production, but also to quantify such changes through the application of defensible methodologies. Minns emphasized the importance of the duration and timing of the use of habitats by fish, and that transient use by fish does not diminish its importance or significance. He also recognized that fish productivity is an outcome of total ecosystem productivity.

Thus the complexity of the issues provided challenges not only to the scientific community but also to those responsible for the management of fish habitat under the *Fisheries Act*. Levings et al. (1997*a*) revealed difficulties around the "operational measures of PC" that would necessitate the use of surrogates to PC. This position had been advanced from DFO "National Headquarters" in 1995 – "effects of a project on the physical, chemical and biological components of fish habitat will normally be assessed as surrogates for impacts of productive capacity" (e.g. refer to Fisheries and Oceans Canada 1998). This position indicates that this is not only possible to do but that it has both validity and accuracy. Validation of these assumptions remains an operational and scientific challenge, and since the initiation of the Habitat Policy, Departmental resources have been, and are being, directed towards their resolution. While there is a well-established array of measures and indices for fish at the individual, population and assemblage levels of organization, validation of measures of productive capacity (or surrogates thereof) and their link to ecological functioning of ecosystems and fish population dynamics remain significant challenges. A perspective on these endeavors and developments, and the need for new information and practical techniques and methodologies, can be obtained from the results of a number of Departmental and other workshops (1996-2004), together with published information. The following comments summarize some of that information.

Research and management needs: Research needs for fish habitat management 1996: Levings et al. (1997*b*) summarized the results of a workshop held in 1996 that addressed aspects of Habitat Management and scientific requirements. The deductions and opinions presented at that workshop provide a reference point ten years after the initiation of the Habitat Policy and therefore reveal the issues facing practitioners and scientists. At this workshop ten "themes" were identified which, in ranked order, were: technology transfer; mitigation, restorative, and compensatory techniques; ecosystem research; technology for habitat characterization; modeling of productive capacity and advancing the understanding of the concept; habitat mapping; research on habitat alterations; retrospective analysis; and risk analysis. It was also deduced that contemporary needs will demand consideration of "protected areas, cumulative impacts, sustainable productivity, environmental quality, and biodiversity: a broad-based "ecosystem approach" was considered to be essential (Levings et al. 1997*b*).

Metikosh (1997) emphasized that there was no consistent scientifically-defensible quantitative method to determine productive capacity and assess habitat quality, and that the links between habitat quality and quantity were absent, as were quantitative methods for assessing the effectiveness of compensation.

Scruton et al. (1997*a*, 1997*b*, 1997*c*) stated that models were available which related salmonid (Atlantic salmon and brook trout) densities to various habitat attributes with a potential to predict changes in biomass and densities. However, these authors also recommended long-term monitoring and modeling projects, evaluation of regional models against a spectrum of habitat types and biological assemblages, habitat manipulation studies (including fertilization), determination of relative habitat importance at micro- and macrohabitat scales, and improvement of microhabitat models for in-stream flow assessments. Amiro (1997) provides a cautionary note when he states that "models (mathematical and statistical representations of population response to physical and chemical environments) that predict the suitability of habitat to support specific fish species are not as valuable – but more readily derived – as models that predict fish populations." Whereas population estimates may be standardized to habitat suitability indices the conversion of the latter to the former is "not trivial."

Gordon et al. (1997) emphasized that the assessment of PC must be without reference to the population level because of the numerous factors that can influence populations, and that all habitats for all "species of value" should be considered that would encompass levels from the site-specific to the watershed. As others had mentioned at this workshop (Levings et al. 1997*b*), surrogates or measures of PC must be practical, simple, and quick to use.

Welch (1997), in reference to research needs in the Arctic, provided comment on contrasting situations where the level of knowledge of food web structure and productivity of important species is rudimentary to non existent (hence basic ecosystem research is valid and essential), as in coastal systems to small Arctic lakes where there is "a good understanding of basic ecosystem structure, energy flow, and physical characteristics" and, because of this, research questions can be more accurately formulated.

Of fundamental concern was the time frame within which research can respond to these recommendations, and it is obvious that short-term funding is inadequate to meet the needs outlined here.

Research needs for fish habitat management 1997: Lester et al. (1997) provide the results of discussions on science for fish habitat management, and the then current directions in fish habitat management, and the need for better "management tools."

Randall (1997), for example, provided a synopsis of work on the field validation of "habitat indicators" of productive capacity as a prerequisite to assessing the impacts of habitat alteration. Assessments were based on habitat inventories and the assignment of habitat suitability using a composite of indicators (links between the indicators and fish utilization and productivity must be defined and scientifically defensible). Cover, depth and substrate types (which may be mapped) were considered to be the key generic indicators.

Area-based habitat plans were considered to be important in the context of site-specific development activities, and although an assessment of the significance of impacts on habitat is closely linked to fishery objectives it was recognized that fish-habitat linkages were poorly understood. Fish-habitat links have been investigated through the use of uni- and multi-variate statistical methods (Randall 1997). For this purpose, different fish metrics were used individually or collectively as "dependent surrogates of fish production" (species presence/absence, species richness, density, fish size and biomass), and links were discerned between fish use of nearshore habitats and the habitat features (such as substrate type, cover, exposure to wind (i.e. fetch), nutrients (phosphorus), and temperature). Although the precision of the relationship models was deemed to be only "moderate," they were being refined to be useful to habitat managers. In this regard, Randall (1997) stated that the field validation of habitat indicators in nearshore areas of the Great Lakes would continue to be a research priority.

Research needs and recommendations from this workshop included the need for large-scale experiments on the effects of habitat alteration to validate decisions and, for example, compensatory measures, and scientifically-based guidelines and tools for management were considered to be a priority.

From a management perspective, it was recommended that innovative compensation agreements should be used to gather data on habitat alterations.

A clear framework for the application of data is needed before designing and conducting inventories, and that in terms of time and space these need to be matched with the scales of development activities affecting fish habitat. In this regard, inventories are needed in support of research and management even though there may be a difference in scale between the gathering of scientific information and those at which most habitat management decisions are made.

It was concluded that habitat management is hampered by unclear objectives, insufficient scientific knowledge, a need to better understand habitat linkages among species to predict consequences of habitat change, and that fish community objectives are needed before the harmful effects of habitat changes can be evaluated (Lester et al. 1997).

Recent workshops in relation to Habitat Science and Management 2002/2003: The results of two workshops provide additional and relevant information on the topic of habitat management, the need for information and its appropriate application in decision making. In 2002 a workshop was held to address critical habitat for aquatic species at risk (SAR) (Randall et al. 2003), and a science technology transfer workshop was held in 2003 as an aid towards improving fish habitat management through the provision of scientific findings (DFO 2004). These workshops provide more current information but at the same time reiterate the concern over the need for basic information and quantitative tools that will facilitate decision making over issues of fish habitat management and the NNL of productive capacity.

It is interesting to note that many statements made almost a decade earlier on the scientific requirements around the Habitat Policy and its application are still valid at the present time. This is related not only to the complexity of the issue but also to the resources directed towards the topic.

Critical Habitat and Species at Risk 2002: An objective of the SAR workshop included the identification of science-based methods for measuring critical habitat, the development of performance criteria for methods employed in species recovery plans, a determination of biological and habitat data needed to measure critical habitat and recommended research in relation to understanding the nature and scale of critical habitat (Randall et al. 2003). The topics, perhaps not surprisingly, are similar to those of the previously-mentioned workshop held years earlier in the context of quantifying the productive capacity of habitat (refer to Levings et al. 1997*b*; Lester et al. 1997).

Many of the issues discussed at the SAR workshop have relevance to the understanding of habitat needs and its function in such under-researched areas as northern Canada where habitat change has been, and is, occurring as a result of mining activities. In relation to the value of habitats, it was recognized that not all are of the same quality or functional value to aquatic resources, and that the identification of critical habitat is necessary for the recovery of endangered species. This deduction is valid for all aquatic species and if one is to provide protection for these organisms it is of fundamental importance that the habitat requirements for all life history stages of a species are identified, and those of the supporting ecological system, over time. Such an approach has fundamental roots in the logical understanding that habitat (which includes physical, biological and water quality aspects because all affect survival and individual and population viability) loss or degradation leads to individual-level effects and population declines. Intuitively, if degradation of habitat occurred for example, during extraction of non-renewable resources, knowledge of the significance of the affected habitat would facilitate decisions regarding the implications of that loss and, importantly, the potential to appropriately mitigate that change or loss and to compensate for it.

Information requirements: It was recognized that quantitative measures of critical habitat are dependent upon the availability of information and that when information is scarce only qualitative results may accrue. In contrast, where the information base is high, quantitative population targets may be obtained as well as a greater resolution of critical habitat. In the latter case, knowledge of population density, life stage growth and survival rates and productivity will allow for the application of models for identifying critical habitat (Randall et al. 2003). Thus density-fish size regression models may be used to determine area per individual (API, Minns 2003) and hence the habitat area requirements of a population.

Research needs and recommendations: Randall et al. (2003) stressed that in many instances habitat-dependent functions are still poorly known, and that further research is needed to link fish density to habitat. In particular, knowledge of the life-stage-specific rates and functional linkages with habitat will permit the use of population models to determine critical habitat. This will be possible with long-term data sets on fish productivity and demonstrated linkages with habitats; an issue that is typically not apparent for many species of fish in northern Canada, especially in the Arctic. The paucity of information will require the incorporation of uncertainty and the adoption of a precautionary approach to habitat management; comments applicable not just to the identification of habitats that are deemed critical (refer to Minns 1997*a*, 1997*b*; Minns and Moore 2002). The adoption of a life-stage approach to identifying critical habitat was emphasized as well as the need for information on carrying capacity and production (Randall et al. 2003). Experimental approaches (micro to macro), behavioral investigations (e.g. including use of radio-telemetry), and the undertaking of case studies were also recommended.

The evaluation of progress towards quantifying critical habitat and other facets of species recovery programs was presented as an important adaptive management and research strategy. For example, the use of fish density-size relationships to predict critical habitat – valid methodology based on data for *Perca flavescens* from nearshore areas in the Great Lakes (Randall and Minns 2003).

Science Technology Transfer Workshop 2003: At the time of writing this report, the Science Technology Transfer Workshop (DFO 2004) probably provides the most current information on the topic of habitat management and related science. The presentation material and the deductions and conclusions from this workshop reveal the ongoing need for habitat-based research, and the progress that has been made towards the identification and use of scientifically-defensible methods to identify and quantify habitat and its function. Most of the development of methodologies has occurred in more temperate regions and consequently applicability to colder regions such as the Canadian Arctic has, in general, yet to be tested and validated.

Deductions and recommendations: Some of the main findings and comments from this publication (DFO 2004) are presented to indicate some of the common knowledge requirements that are fundamental to meeting the needs of habitat managers irrespective of their location within Canada. But, in general, Habitat Management often deals with site-specific issues of habitat change whereas Departmental Science endeavors to understand fish-habitat linkages from a more "holistic" perspective. Measuring habitat productive capacity, threshold response to habitat alteration, effectiveness of habitat compensation, and managing risk and uncertainty were discussed at this workshop.

Habitat function and general comments: It was stated that habitat function could only be understood by "investigating habitat-specific process rates of different life stages of fish at a population or watershed scale." Accordingly, knowledge is required in the development of HADD criteria in the context of responses at these community and ecological scales. An initial assessment of habitat capacity could be low-resolution, broad-scale maps of habitat productive capacity generated for large areas. But, resilient and sensitive habitat would also be identified at the site level so that more detailed studies of capacity and sensitivity could occur if warranted. Tools for the determination of "broadscale" classifications of habitat were considered to be currently available for freshwater systems.

Specific approaches, habitat surrogates, resiliency, multi-factor analyses and assessments, habitat manipulation, ontogenetic considerations, lacustrine-fluvial linkages: Specific presentations at the workshop exemplified some progress towards understanding the effects of habitat change on fish, and the linkage between habitat and productivity; a topic of concern expressed previously (Lester et al. 1997; Randall et al. 2003).

Randall et al. (2004) revealed the utility of multivariate analysis for mapping habitat productive capacity in nearshore areas of the lower Great Lakes. Fish density was habitat dependent and those with most cover had the highest densities (Ontario lakes). Similarly the nature of physical habitat was correlated with coastal exposure to the extent that seasonal fish biomass (related to habitat productive capacity) could be predicted from knowledge of exposure factors.

The manipulation of habitats and evaluation of impacts were recognized as important to elucidate the responses (linear/threshold) of populations (Mills et al. 2004; Smokorowski et al. 2004; Stanfield and Kilgour 2004), as was whole lake experimentation (e.g. Blanchfield et al. 2004).

Pratt and Smokorowski (2003 provided evidence that structurally-complex habitats contained the highest fish densities in lakes in Ontario, and concluded that "underwater visual methods for determining habitat-specific density estimates, in combination with published production:biomass ratios, provide a simple and effective method for determining habitat-specific production. In this context these authors noted the fidelity of fish to specific habitats in this study (which was an assumption of study design).

On a cautionary note Minns (2004) indicated the need to assess the ontogenetic use of habitats by fish to fully understand links between supply and population response. Minns (2004) revealed how a population of lake charr *(Salvelinus namaycush)* was more sensitive to habitat supply at the juvenile stage than at the spawning stage (a situation that is somewhat contrary to usual expectations regarding critical habitat). He also discussed the value of using habitat area-per-individual (API) assessments, while at the same time recognizing the relative lack of knowledge of the biology of certain fish species, but the advancement of population dynamics and the role that habitat may play therein.

Scruton and Clarke (2004) stated there was a lack of understanding about the linkages between lacustrine and fluvial habitats for fish despite the intrinsic relation to populations. They also considered that habitat use by fish in Newfoundland was unique because of the small number of species that inhabit the watersheds, low aquatic productivity, and an abundance of fluvial habitat and numerous small lakes in watersheds. Individuals within these low-productivity systems do not always display a strong affinity for particular habitats, and large, inter-habitat migrations occur. This situation, of few species and oligotrophic conditions, however, resembles that in more northern regions of Canada. The observations serve to emphasize the problems of generalizing about habitat use that is mediated or controlled by factors that change geographically and spatially (e.g. temperature).

Research needs, knowledge gaps, recommendations and conclusions: A priority for science was the continued investigation of habitat-dependent process rates and productive capacity at the population level. In addition, acceptable time frames for the assessment of productive capacity are needed as well as community targets for native fish species.

The often non-linear response of fish populations to habitat change (perceived as resilience up to a threshold (refer to Pratt 2004) presents a particular challenge to the quantification of effects of habitat change (to identify such threshold responses a long-term commitment to research was deemed necessary (DFO 2004)).

It was also concluded that more research is required on the functional relationships between habitat and the different life stages of fish (habitat-dependent process rates), and because fish-habitat linkages are dynamic, "generalization and extrapolation of habitat capacity remains risk adverse" (DFO 2004).

A number of "knowledge gaps" pertinent to this report included the role of wetlands, guidelines for compensation monitoring, determination of compensation effectiveness, methods for quantifying habitat, determination of the function of different habitats, an expansion of experimental habitat removal projects, studies of habitat dynamics (spatial and temporal variation in habitat use by fishes), the identification of, and the need for, guidelines for identifying the appropriate scale for measuring the success of compensatory measures.

Recommendations from this workshop included the need to establish a national habitat-science advisory working group to promote annual collaboration and communication; a workshop to establish consistent inter-region benchmarks for the determination of no net loss; establishment of GIS-based inventories of habitat supply, linking with the habitat referral tracking system to deal with cumulative effects; implementation of risk assessment for referrals; provision by science of methods for determining first-tier maps of habitat capacity and sensitivity; adoption of adaptive management for monitoring compensation success; and the development of operational guidelines for determining HADD.

Assessment of productivity and production, habitat capacity and surrogates: Randall (2003) provides an excellent and timely commentary on the determination of productivity and production, habitat capacity and productive capacity, biological indices of production and biodiversity, habitat surrogates of productive capacity, and population geography.

Randall recommended that habitat assessment should be linked to fish populations so that the impact of habitat alteration is explicitly evaluated at the appropriate geographical scale. He also commented that the measurement of productive capacity using biological indices or habitat surrogates is based "on the sometimes untested assumption that these static indices are indicators of the dynamic population processes of recruitment, survival, and growth that together determine production." Accordingly, validation of the habitat surrogates is needed at the population level, and, similar to the comments made by others, he considers the link between stock and recruitment processes to habitat (i.e. total environmental conditions) for the various life stages.

Rabeni and Sowa (1996) report that a general feature of habitat evaluation models is that of an equation linking some independent habitat variables to a single output (dependent variable) about habitat quality or a fish characteristic. They contend that models with habitat quality as an output variable (many habitat quality indices) have no direct relationship to fish other than the implied one that as habitat quality increases then so do certain attributes of the fish population of interest. Of more relevance are models relating fish characteristics with habitat features. They provide the example of multiple-regression models to predict density and biomass of smallmouth bass based on habitat features. Of 5 initial variables, only 2 (maximum summer temperature and percent pool area) significantly contributed to final models. Contrary to expectations, in no instance was cover associated with abundance – a contradiction with the results of microhabitat studies where cover was important in stream reaches and the distribution of the fish (Rabeni and Sowa 1996). Microhabitat variables (such as water velocity, proximity to cover, depth, and substrate) are often used to construct habitat suitability relationships for fish. Medium scale (at the riffle, run, or stream-reach scale) indicates the variety of microhabitat conditions available. Landscape-scale considerations take into account large-scale movements of fish and effects of the physiographic variables and land use (Lewis et al. 1996). These comments are further reinforced by those of Hartman et al. (1996) who consider that it is an oversimplification to attribute population limitations to one factor because fish respond to their total environment. Several physical and biotic factors acting alone or in concert can potentially limit production (Bisson 1989). They shape habitat preference, determine patterns of behavior, and ultimately affect annual production levels.

Hubert and Rahel (1989) described simple and multiple regression models which were the first quantitative descriptions of relations between abiotic features and the biomass of white suckers, creek chub, and longnose dace. The models accounted for a substantial amount of the variation in standing stock without consideration of other factors that influence their abundance such as community composition and hydrological phenomena such as floods and droughts. Hubert and Rahel contend that the inclusion of these other parameters would account for more of the variation in standing stock, thereby endorsing the use of multivariate models. Yu et al. (1995) also endorse the use of logistic regression techniques in elucidating relations between several environmental factors and their effects on habitat use by fish. They also contend that multi-year studies are required to characterize habitat preferences because habitat suitability criteria developed from a single year of data may not reflect the conditions to which fish may be exposed (Yu et al. 1995).

Thus there continues to be a need for the validation of "tools" for indices and surrogates of habitat productive capacity, and to link them with population and community fish production (Randall 2003).

Net change equation: Determining or predicting fish productivity may occur by direct measurement of production rates of all species, measurement of biological indices such as biomass and surrogate habitat variables (Minns 1997*b*). Furthermore a reference condition is required from which or upon which comparisons may be made to reveal changes in productivity and hence ecosystem health over time. As Minns (1997*b*) stated, the reference point allows judgments to be made over such activities as restorative measures for fish habitat.

Minns (1997*b*) presented a "net change equation" for assessing the no net loss of the productivity of fish habitat that lies between the positions of no development and conservation. As such, his approach provides for quantitative assessments to be made, which are preferable to non-science-based decision making. However, the importance of local knowledge is recognized and not discounted. Underpinning the approach is the expectation that effects can be quantified. While this may be so for certain circumstances, in those situations where information is inadequate and therefore the outcome of decisions uncertain, Minns (1997*b*) suggests the application of the precautionary principle. By doing so, sustainable fisheries may be "reestablished" despite uncertainty. Through the use of precaution, the onus is placed on the proponents of habitat change to prove that there will not be a loss of the productivity of habitat, or that their actions may be mitigated or compensated for. This, as stated earlier in a similar context above, is only possible and likely successful if the information supporting such actions is adequate, and the methodologies employed are quantitative and defensible.

The quantitative scheme devised by Minns shows how net change balance may be computed, and at the same time supports the use of relative indicators of productivity like the Habitat Suitability Index (HSI, US Fish and Wildlife Service 1981). In the practical sense Minns (1997*b*) outlined the steps taken to follow such a "defensible method" with reference to the Great Lakes. This includes the identification of life-history stages of species considered and the generation of habitat-suitability models for them. Habitat features such as depth, cover and substrate, are also determined and using distribution data, species lists are compiled and grouped according to, e.g. thermal requirements, trophic levels, etc., and suitability matrices developed. Using GIS-based estimates of areal supply and combinations of habitat features "weighted usable areas (WUA)" are computed for each fish group and life cycle stage. It is apparent that such methodology appears quite simple if one has the requisite information base and resources to provide assessments and validation. Murphy (2001) reported the results of the application of a Habitat Alteration Assessment Tool (HAAT) using the protocol developed by Minns et al. (1995). The HAAT model organizes the habitat features (i.e. depth, substrate and cover) and parameters assigned by the user into a composite suitability matrix for each fish species and life stage combination (Minns et al. 2001). The model then calculates weighted suitable areas (WSA) by multiplying each habitat area by a predetermined suitability value. Suitability values are computed as part of the HAAT model using a detailed database of fish habitat requirements for all Great Lakes fish species. The WSA is calculated for the pre- and post-development scenario, and the values for each are compared to determine net gains or net losses. Non compliance with monitoring requirements was found for the majority of projects sampled, and this provides an indication that similar non-compliance issues can be expected for larger projects, where the potential for using HAAT to determine impacts associated with development applications exists. As such, without the effective use of monitoring strategies, and because of non compliance with monitoring requirements by project proponents, assurances that compensatory initiatives are satisfying sustainable development objectives are compromised (Murphy 2001).

Minns (1997*b*) mentions the inadequacy of effort to link the supply of habitat to fish population dynamics, and the obvious relationship that exists between whole-ecosystem productivity and the inherent link to fisheries. He recommended research that includes the development and validation of practical habitat management tools for site assessments and habitat-area mapping, population and community models and estimates of habitat supply, spatial and temporal distributions of productivity and validations of such, and the development of new measures.

The use of "resource selection functions" (e.g. Jones and Tonn 2004) to describe species-habitat relationships have been considered to add more scientific rigor to the relationships than do HSI models. Resource selection functions are mathematical functions that are proportional to the probability of an area being used by an animal (Manly et al. 1993). They rely upon specific data and can address multiple variables. The gathering of such data is often difficult, but in the shallow, clear waters of the Canadian Arctic (that permitted visual observation of fish), Jones and Tonn (2004) successfully investigated habitat use by Arctic grayling through the application of resource selection functions. Determination of habitat use by two age classes of Arctic grayling in an upstream, natural habitat then permitted comparative assessments of habitats used in the artificially-created Panda Diversion Channel (that restored connectivity among lakes in the watershed used for diamond mining and hence allowed for fish migration and other uses). The application of resource selection functions permitted Jones and Tonn (2004) to describe habitat use by Arctic grayling in Barrenlands streams. They could not use the approach for assessment purposes because of the lack of data on the specific streams that were eliminated prior to the construction of the Panda Diversion Channel, however, they demonstrated the value of the use of quantitative measures in the determination of habitat use models, which, when associated with other research in this location (Jones et al. 2003*a*, 2003*b*, 2003*c*) significantly advanced knowledge of the ecology of Arctic grayling. They also revealed the low likelihood of short-term success of this artificial habitat stream in the Arctic as compensation for that which was lost, which, in itself provided valuable information to guide future decisions regarding compensation and restoration of fish habitat in this region.

Consideration of uncertainty and complexity: Minns and Moore (2002) discuss uncertainty and complexity in decision making with respect to habitat management and the assessment of the productive capacity of fish habitat. Their position is exemplified through the use of a quantitative framework for assessing the NNL of lake habitat in the Great Lakes. The approach assessed how uncertainty affects the net change results that Minns (1997*b*) recommended as a means to assess changes in habitat productivity. Using case studies and simulations based on a net gain or loss of productive capacity, these authors concluded that despite broad uncertainty of fish-habitat relationships, uncertainty can be incorporated and provide for "robust decisions." Their assessments revealed the need for habitat compensation (in like-for-like circumstances) to be at least double that affected, to incorporate uncertainty and satisfy NNL. They also emphasized that attention to minimizing loss and maximizing any offsetting created habitat, is of prime importance. Thus, the need for quantitative evaluations of compensatory habitat productivity over time to reduce uncertainty is identified as a logical extension of the deductions and conclusions of Minns and Moore (2002).

Species-specific production rate per unit biomass ratios: Hubert and Rahel (1989) stated that fish density cannot be taken as the sole indicator of habitat quality and that it needs to be examined in relation to the mean size of fish. Taken together these parameters are indicators of both biomass and production. "The production/biomass ratio (P/B), when measured at the steady state is an index of the energy in the biomass relative to the energy necessary to support that biomass. It is a measure of the effectiveness with which energy is stored and may be applied to either a species, population or an ecosystem as a whole" (Johnson 2002).

Randall and Minns (2000) discuss the merits of using species-specific production rate per unit biomass (P/B, per year) ratios in the derivation of Habitat Productivity Indices (HPI). They base their approach at the population level and consider that fish production is the best indicator of the "quantitative performance of a fish population in any type of habitat and it is a measure of productive capacity." For any period of time fish production is the product of average biomass and specific growth rate, hence fish production and fish biomass are correlated. The approach presented by Randall and Minns (2000) applies to populations that are not increasing or decreasing in abundance with time, and for lakes the specific production was inversely related to the maximum size of the fish and positively related to lake productivity (most P/B ratios were also within this range for fish-inhabited rivers (Mann and Penczak 1986, cited in Randall and Minns 2000). The utility of this index is emphasized through its relationship to population dynamics wherein maximum P/B is equal to the maximum rate of population increase. Both expressions of growth are linked to the productive capacity of the habitat. Thus the rate of population growth, and accordingly P/B can be used to estimate "the amount of time required for a population to establish itself after a catastrophic decline, to colonize a new habitat, or to withstand exploitation" (Randall and Minns 2000).

A habitat productivity index was proposed as a measure of habitat suitability and productive capacity (based on the assumption of a correlation with average fish biomass and that biomass is linked to habitat productivity). Randall and Minns (2000) assessed the validity of the HPI and found that although both biomass and fish size were significant predictors of HPI, the latter also accounted for fish size structure. Accordingly, HPI was considered to be a more "direct fish assemblage measure of habitat productive capacity than biomass alone" (Randall and Minns 2000). The authors caution, however, that this hypothesis was, at that time, yet to be tested in an area where fish production has been estimated in the littoral zone of a lake, and that the use of HPI as an index of productive capacity required further validation in relation to populations and community production.

Hayes et al. (1996) recognized that the dynamics of fish populations are the ultimate measures of the quality and the quantity of fish habitat, as growth, survival and reproduction are directly related to the requirements for specific abiotic and biotic resources. Hayes et al. (1996) state that there is no single "best" measure of the impact of habitat alteration on a fish population and that habitat alterations do not always affect all population characteristics to the same extent. These authors provide the example where small changes in recruitment can substantially change carrying capacity, while producing a small change in "surplus" production of the population (i.e. that which may be harvested in a sustainable manner). This, as recognized by others (e.g. Minns; Randall), means that the impact of habitat alterations needs to be evaluated across several population-level descriptors.

Hayes et al. (1996) commented on the unexploited populations and deduced that the "equilibrium density" is probably the most important population characteristic to compare if the goal is to increase abundance.

Morphoedaphic index, predictions and estimates of fish yield/assemblages in lakes: Assessments and predictions of fish (species, numbers, biomass, etc.) within lakes will facilitate decisions regarding the relative acceptability of those lakes that could be eliminated or changed due to industrial development (e.g. for waste disposal and mining). This is notwithstanding the need to know the functional role of a lake(s) within and among watersheds.

Multivariate analytical techniques have been, and are being, used to relate habitat changes to aquatic organisms (e.g. Kelso 1996; Randall et al. 2004). Simple indices were derived previously that related fish productivity to lake composition. For example, Northcote and Larkin (1956) indicated that a good correlation existed between total dissolved solids and lake productivity between values of approximately 50 and 250 mg/L. An expansion of this relationship has occurred and the simple morphoedaphic index (MEI) has been used globally as a predictor of fish yield from lakes (Ryder 1982). It is considered to have its greatest utility in the "regional" context wherein mean climatic conditions are similar (Schlesinger and Regier 1982). Fish yield is considered to be that harvested and as such, is a substantial and predictable portion of total internal production (i.e. productivity). The following comments indicate the concept of the use of MEI and some of the possible limitations. Its utility may be questioned depending on the circumstances to which it is applied. However, through the gathering of data prior to the elimination of lakes and the application of them to information gathered due to the removal of fish, opportunities exist for the index, and other analyses (Oglesby 1982) to be assessed using precise data.

Ryder (1982) stated that the MEI spans the gap between ecological theory and management application and results in a compromise in a partial loss of ecological rigor. The MEI has been lauded and criticized, but its utility depends upon the nature of data included in the analyses. Ryder (1982) stated that numerous factors influence the productivity of natural lakes, and that they may affect each other also and collectively influence productivity and hence fish yield. In Ryder's words the MEI "combines methodological simplicity with conceptual complexity." Oglesby (1982) considers the MEI to have a fundamentally-sound theoretical basis and comments that fisheries scientists have seldom critically evaluated the abiotic factors that control production in inland waters.

The MEI was developed to determine empirical relationships between fish yield and abiotic factors, to provide fishery managers with an easily-applied technique, and a global synthesis of production processes in aquatic systems (Ryder 1982). Inherent in this index is that energy entering the aquatic system is retained because of its morphology. The basic MEI (N/mean depth), where N represents a nutrient value or correlate (total dissolved solids (TDS) was used in regional models of MEI but, for example, phosphorus could replace TDS where it is considered to be limiting). According to Ryder (1982) the concept may be applied at different trophic levels to include such measurements as chlorophyll *a*, quantitative measurements of algae, zooplankton or benthos. Ryder (1982) explains the concepts behind the MEI, and the role of nutrients, energy, morphology, depth, volume, and area are discussed.

In large lake basins the morphometry is such that it channels, constrains, dissipates, and sinks energy and nutrients disproportionately to that of small lakes where availability of nutrients may be the single key factor in the regulation of fish production (Ryder 1982). Thus constraints on fish production in small and large lakes differ, however, mean annual temperature and area have significant correlation with fish yield on a global basis.

In lakes within a watershed in which geological, physiographic, and soil conditions are approximately equivalent, Ryder considers that it should be possible to predict fish yield using mean depth (assuming that there is variation among the lakes in this variable). Similarly, lakes receiving different levels of nutrients but with similar mean depths should be amenable to predictions based on nutrients alone or some surrogate thereof. For example, Ryder (1982) states that for a series of lakes of less than 6-m depth, good correlation was obtained between chlorophyll *a* and fish yield (Jones and Hoyer 1982).

Thus on the global scale temperature and area are of prime importance. At the regional level where homogeneous climatic conditions exist, mean air temperature may be considered constant and area also, nutrient input and lake morphometry are primary factors and at this level the MEI has the greatest utility (Ryder 1982; Schlesinger and Regier 1982).

At the smaller infra-regional level where either nutrient loading or morphometric characteristics (such as mean depth) may assume constancy, fish yield may be predicted based on mean depth or nutrient concentration, respectively (Ryder 1982). Oglesby (1982) extends the concept of use of MEI by suggesting such variables as total fish yield, area, depth, hydraulic retention time, littoral:limnetic ratios, edaphic factors such as bedrock type, land use, soil type, waste output, as well as TDS, specific conductance, winter total phosphorus, water chemistry, turbidity, and biotic factors such as phytoplankton standing crop in the epilimnion, annual primary production, and standing crops of zooplankton and benthos. While the list is extensive there is also recognition the verification of MEI-type predicted yields should occur, possibly through whole lake studies such as those at the ELA (Schindler 2001); an approach that has also been suggested in relation to assessing the responses of aquatic communities to changes in their habitat (Kelso et al. 2001).

Studies on the capacity of lakes for producing sockeye salmon have been carried out over decades, and models have been produced that relate the rearing capacity to carbon production (Williams et al. 1997). A strong correlation between maximum smolt biomass and lake carbon production has been used to validate the use of the photosynthetic rate model, which predicts rearing capacity. The model relies upon much data, and is seemingly simple to use.

A similar simple model is used to guide the stocking of lakes in BC. It is an expansion of the MEI concept and relies on an assessment of the littoral regions of a lake. The model is based on the research of Stringer et al. (1980), who used in their analyses a wide variety of lakes that differed in physical and chemical characteristics. They considered that the use of shoreline length as an index of carrying capacity for young rainbow trout was questionable and that the use of shoal area (<6 m depth) was a more useful factor to consider. But, overall lake production was predicted through the use of both shoal area and surface area: Shoal area (P_1) + Surface area (P_2) = Total Production. P_1 = kg/ha/y which equals 10xP_2 (Stringer et al. 1980). These authors considered that estimates of overall productivity need to be obtained from the components of shoal production (benthic), and production associated with the water column if a suitable means of comparing lake systems is to be obtained. However, their findings have utility in relation to decisions to manage and assess the productivity of lakes, to use lakes as compensatory options regarding changes in productivity, the stocking of lakes, and their creation for the purposes of compensation. It is not known whether their results have applicability to the lakes in the colder regions of Canada, where littoral and shoal areas typically have lower productivity due to the duration and thickness of annual ice cover.

Williams et al. (1997) recognized the complex and difficult tasks of integrating approaches to address ecosystems, and deduced that there are two divergent views. The need to construct more detailed models to fully understand the dynamics of production in different geoclimatic zones through long-term research, and the need for management to have practical "tools" to address immediate needs. For the latter, the maintenance of spatial and structural diversity drawing upon the link between habitat area and production has merit, but, they caution, the former cannot be ignored.

Marshall (1996) provided an example of the use of assessment techniques that address ecological properties that are essential for the survival and production of lake trout. The concept draws upon the need to evaluate critical habitat and understand fundamental habitat requirements that have the potential to limit or control production. The relevance of the example lies in its potential utility when addressing lake destruction and compensatory and restorative opportunities.

The following comments are from Marshall's (1996) publication. The ecological properties essential for the survival of lake trout can be placed in a hierarchical framework. Marshall commented that prior to any consideration of yield potential, it must be determined that the lake is capable of supporting a self-sustaining lake trout population (with surplus production available for a fishery). Initially, the lake is plotted along a continuum of mean depth and primary productivity to evaluate whether the critical temperature and oxygen requirements of lake trout are met. A review of other physical habitat attributes essential for the success of the population follows, such as the availability of spawning substrate and nursery refugia. If the fundamental environmental and physical habitat needs are met, a viable lake trout population could be expected, and an evaluation of harvest potential is appropriate. An estimate is made first of total community yield, then of the sustainable harvest of lake trout alone, applying thermal habitat volume and lake area models. Finally, adjustments are made to this estimate through consideration of modifiers that may constrain lake trout production, including community biodiversity and various anthropogenic stressors. Small lakes (<1,000 ha) are most affected by these constraints.

According to Marshall (1996) if a lake is to support a self-sustaining population of lake trout, it must be suitable with respect to the primary environmental determinants that control or limit the species (Fry 1947; Ryder and Kerr 1989). This can be established through application of the oxygen depletion models of Ryan and Marshall (1994). Lakes that experience a deficiency along one of the environmental axes would be unsuitable or marginal for lake trout in their current state. Unless conditions were improved, their future would be uncertain with regard to supporting a naturally-reproducing stock of lake trout, let alone a sustainable fishery.

The next step is to determine whether other essential physical elements are in place for lake trout. Lake trout have very specific spawning and nursery requirements, and they can be readily degraded. If this critical habitat is in short supply, reproductive success and early age survival could be greatly diminished, and a fishery for naturally-produced fish could not be expected. If both environmental attributes and physical habitat are sufficient for lake trout, total community yield should be estimated to establish an upper bound to lake trout yield. A more precise estimate of potential lake trout harvest may then be made through application of the thermal habitat volume (THV) or lake area model.

Marshall (1996) stated that lake size is a primary consideration, as species biodiversity can vary substantially in small lakes (not necessarily so in Arctic lakes though), especially those of surface area less than 1,000 ha. Such lakes are exceedingly important, as they comprise more than 80% of Ontario's 2,000 lake trout waters (Martin and Olver 1976). If the fish community is simple, with a planktivorous lake trout population, a yield equivalent to the THV or lake area model might be expected; if not, resources must be partitioned among more species, and yield expectations for lake trout in these small lakes should be downgraded. Human activities within the lake's catchment must also be assessed, as there are a number of potential stressors, besides fishing, that could contribute to a reduction in lake trout yield from the model prediction. As a general rule, small lakes would likely be more vulnerable to these cultural impacts and their lake trout populations would be more at risk of depletion and even extirpation triggered by the additional stress of exploitation. A cautious approach to yield estimation is advised when these conditions are encountered.

Restoration of fish habitat: The successful restoration of habitats relies upon our knowledge of ecology; it is Bradshaw's (1996) "acid test," wherein restoring ecosystems tests the state of science and ecology (Kelso et al. 2001). Through restorative actions it is expected that an ecosystem will return to a close approximation of its condition before it was disturbed (Pastorek et al. 1997). To guide such actions examples are required, but cause-response information for ecosystem-based restoration is limited and sorely needed (National Research Council (NRC) 1992, cited in Imhof et al. 1996).

The restoration of aquatic habitats relies upon information that is also needed for basic habitat and fish management. Thus comments relating to compensation and the linkages between habitat and fish are as applicable to restorative activities as they are to compensatory measures. In each instance ecological knowledge is fundamental to understanding the complexities of species requirements and links with their biotic and abiotic environment. The publication edited by Kelso (1996) describes restorative efforts, the complexity of the issue and its quantification.

Pastorek et al. (1997) described an ecological planning process for restorative projects. Within the plan they emphasized the importance of defining objectives related to the appropriate ecosystem structure, function, and ecosystem scale; the role of ecological model restoration hypotheses, and key ecological parameters; explicit considerations of uncertainties considered in design; guidelines for project design and feasibility analysis and the use of experimentation at this stage; and monitoring and adaptive management after implementation of a design. These authors considered that certain ecological perspectives should guide environmental restorative projects. They considered that a "key species functional perspective" (in addition to physical habitat features, presence of certain organisms and their functional role) drives the outcome of a restorative project.

Because ecosystems consist of a mosaic of habitats that varies with space and time, and restoration objectives are seldom met exactly, spatial diversity and flexibility in design are required. In this regard, Kemp et al. (1999) stated that river reaches of uniform depth had low habitat diversity compared with reaches with shallow and deep areas, and that fish species diversity was highest in physically-heterogeneous channels. They considered that the development of depth and velocity occurrence matrices for each functional habitat is an important step towards the goal of being able to predict the result of physical modifications on habitat frequency in river channels, whether they be changes in channel shape or discharge pattern. The likelihood of the retention of habitat diversity was greater if the channel retained its natural width and was morphologically variable. Management of such compensatory and restorative measures must be adaptive and implemented in a tiered manner such that early results might guide the implementation of successive actions (Pastorek et al. 1997). Although adaptive management requires flexible goals and designs, a long-term commitment to detailed monitoring and fine tuning after initial implementation are fundamental requirements.

When consideration is given to the success of restorative measures for fish habitat, reliance upon the presence or absence of a particular species over time to indicate its success, is fraught with dangers. This is because in this instance a detailed knowledge of ecological function is not required (Bradshaw 1996). For example, the species may persist only for years and later disappear because the underlying functions had not been addressed. It is crucial, therefore, that the development (restoration) of the ecosystem be upwards and along an unobstructed path related to structure and function. Furthermore, Bradshaw (1996) advises caution when taking any character as a surrogate for all of the development processes (a topic that has been recognized by many others, e.g. Hartman et al. 1996; Hartman 2004).

To address the issue of restoration success, experimentation is required that includes comparison, which is an essential ingredient of scientific ecology (Bradshaw 1996). It is important to recognize, therefore, that restoration is a progressive process and as such, criteria defining success are difficult to decide upon. These performance indicators (i.e. structural or functional elements of the ecosystem that are to be used to judge the success of the project) should be linked to project objectives within the known attributes of regional reference ecosystems, and it is only logical that multiple performance indicators should be used to minimize the risk of missing important ecological effects (Pastorek et al. 1997). It is important that project designs incorporate a whole-ecosystem perspective for "ecological restoration is an holistic approach not achieved through isolated manipulations of individual elements but through approaches ensuring that natural ecological processes occur" (Kauffman et al. 1997, cited in Hartman 2004). Furthermore, it is apparent that the time frame over which to judge success is most likely many years (Reisenbichler et al. 2003); Bradshaw (1996) gives examples of 10-year time frames. Lester et al. (1997) estimated that for a study designed to detect a two-fold change in population abundance, 7 years before and after the change would be required; longer than many present studies. In a broader sense, Kelso and Wooley (1996) suggest that society's ability to protect and rehabilitate will be least where scientific uncertainty and sampling variation are greatest. Notwithstanding these concerns Bradshaw (1996) proposed that natural processes should be used wherever possible and that following the removal of "degrading influences" this may be sufficient.

MINING DEVELOPMENT AND ASPECTS OF COMPENSATION AND RESTORATION

There has been significant effort by industries and regulators to mitigate the impacts of development at a variety of scales of potential impact, to compensate for effects that cannot be fully mitigated, and to promote the restoration of fish habitat upon termination of the development.

Aspects of compensation for damage to fish habitat are presented here in relation to the three mining sectors examined in this report. In addition, historical and current information on the elimination and degradation of lakes and streams for mining in Canada are presented to reveal the scope of the activities, the requirement for compensation, and the policies and legislation under which they were permitted.

Diamond mining

The Ekati diamond mine (BHP Billiton, BHPB): According to the authorization issued to this diamond mining company, "To compensate for the loss of fish habitat associated with the Project, BHP Minerals shall implement the Fish Habitat Compensation Agreement negotiated between the Department of Fisheries and Oceans (DFO) and BHP Minerals." In accordance with this authorization which recognized that "The Project will impact directly on 12 lakes within the claims block" BHP placed $1.5 million into a Habitat Compensation Fund established and administered by DFO. The fund was to be used to support habitat compensatory projects (BHP also constructed the Panda Diversion Channel that diverts drainage water around active open pits and also provides stream habitat as part of the compensatory measures for development impacts on streams).

The $1.5 million are being allocated with assistance from an Aboriginal Advisory Committee (AAC) formed of the five Aboriginal groups that were most affected by the diamond mining development. Plans for projects were developed and assessed by Public Works and Government Services Canada for their acceptability under the terms of the contract notice, then by DFO and the AAC. Projects must meet specific criteria that include an understanding of the scope and importance of the requirements of the Habitat Compensation Fund. The proposals are to be community-based projects within or near the affected Aboriginal communities, and within a defined geographic area. The proposed project, and its methods and controls must demonstrate an ecosystem approach, benefit multiple species, set a variety of resource management objectives, and address concerns raised by traditional resource users and communities in the defined geographical area. The level of effort must be scaled to the project, and there must be efficient use of equipment and personnel. Potential problems and methods of handling them need to be identified. Techniques and approaches that can be applied successfully elsewhere should be developed. The quality of design, original ideas, and sound, feasible procedures are to be reported. Public involvement is encouraged through consultation, and incorporation of communities through employment or training at all stages of the project (concept/design through to monitoring). Partnerships with private companies and/or government departments are also involved.

At the time of writing this report in 2004-05, none of the projects funded by the Habitat Compensation Fund specifically addressed diamond mining development, but all were associated with fish habitat. For example, $500,000 was allocated for the restoration of fish habitat at Matthews Lake, at the former Salmita gold mine. Fish habitat was damaged due to access road construction, culverts, and in filling. Two-hundred-thousand dollars will be used to remove fish migration barriers along an Arctic coastal stream in the Kugluktuk area, to provide access to a lake for a diminishing population of charr. Seventy-thousand dollars have been allocated to identify potential habitat issues in a lake with declining in fish populations and which was subject to "over fishing." One-hundred-ten-thousand dollars were allocated to determine if the decline in fish quality in Stark Lake is habitat related.

The remainder of monies within the Habitat Compensation Fund ($500,000-$700,000) was to be directed at community-based habitat restorative projects. A "Habitat Works Database" of candidate projects for the fund, and for future compensatory projects is being developed with the Department (J. Dahl, Area Chief, Habitat Management, DFO, Yellowknife, NWT; pers. comm.).

Diavik (Diavik Diamond Mines Inc./Aber Diamond Mines Ltd.): The components of the compensation plan related to the Diavik mine include the Lac de Gras pit area, which, at the conclusion of mining, is to be restored, and one stream that will be enhanced on the site (the plans for both are currently under review). An "on-site" lake is to be enhanced, and an additional lake is to be included in the compensation plans which have yet (as of December 2004) to be submitted for review (J. Dahl, Area Chief, Habitat Management, DFO, Yellowknife, NWT; pers. comm.).

Snap Lake (De Beers Canada): A rock reef (intended for spawning fish) has been constructed in Snap Lake as compensation to offset the damage to a small lake due to the development of a processed kimberlite containment facility that occurred during the advanced exploration phase of the mine development. It is expected that the reef will add to the complexity of habitat in the lake and provide a spawning substrate for fish. Criteria for the success of this reef include the stability of the structure and its "functionality" with respect to the successful spawning and egg development by lake trout. These "success criteria" and a decision flow chart need to be devised to provide quantitative rather than qualitative data that will assist the management and assessment of other projects (J. Dahl, Area Chief, Habitat Management, DFO, Yellowknife, NWT; pers. comm.).

De Beers Canada is also to undertake the enhancement of a stream, and impacts on lakes are to be compensated for at a location that has yet to be determined.

Timing of compensation and related issues: While some fish habitat compensatory works are being carried out at present, others (e.g. in the case of Diavik) are to be effected at the end of mining the current kimberlite pipes and before the company breaches the dyke in Lac de Gras that surrounds the mine pit sites.

According to J. Dahl (Area Chief, Habitat Management, DFO, Yellowknife, NWT), it has been difficult to identify suitable habitat compensatory projects that are also supported by Aboriginal groups. A number of projects in "pristine" lakes that had been identified for modification in the expectation of greater fish production did not advance because of this lack of support. In this regard, while Diavik's plans for stream enhancement and pit shelf enhancement will proceed, island lake enhancement and off-site works are stalled because of community resistance to enhancing undisturbed systems. For similar reasons, plans for the enhancement of biological productivity at the Ekati mine site have also been stalled. Compensation measures at the Snap Lake mine site are, however, to be completed within one year (J. Dahl, Area Chief, Habitat Management, DFO, Yellowknife, NWT; pers. comm.).

Overall, there has been no whole lake replacement and there has been only limited, and "seemingly successful," enhancement that offsets the habitat that has been lost (i.e. Snap Lake reef) (J. Dahl, Area Chief, Habitat Management, DFO, Yellowknife, NWT; pers. comm.).

Research: Detailed aquatic monitoring programs are required of the diamond mining industry to determine changes to the local environment, whereas research *per se* is limited. However, at Ekati (BHPB), there is an ongoing investigation of the Panda Diversion Channel to assess its productivity and suitability as a compensatory structure (e.g. BHP Billiton Diamonds Inc. 2003, 2004; Jones et al. 2003*a*, 2003*b*; Jones and Tonn 2004).

This research information is in addition to the environmental monitoring data generated by the diamond mining industry.

Oil sands mining

The initial removal of oil sands did not impact fish habitat or it pre-dated the present habitat protection provisions of the *Fisheries Act*. However, the current expansion of the industry will impact fish habitat and accordingly mitigation, and compensatory and restorative measures are required.

In July 2004, DFO issued an authorization to Canadian Natural Resources Ltd. (CNRL) that is valid until 2050, to enable the company to conduct works and undertakings in the Athabasca River Watershed. The project comprises the damming of the Tar River (Figure 15) to create a compensation lake (Comp Lake – 76.7 ha) in order to offset the damage to habitat caused by multiple stream diversions as part of an open pit mine plan.

Figure 15. Tar River, Alberta.

Future additional compensatory works will include the construction of spillways, permanent diversion channels (46.7 ha) and channel reconstruction, all designed to provide fish habitat and be geomorphically stable. The company must demonstrate that Comp Lake is functioning as intended before proceeding to the next phase of the authorization. The compensation is intended to offset permanent losses of 302 ha for the Tar and Calumet Rivers and tributaries, and temporary losses in Calumet Lake (70 ha). The habitats impacted are utilized by Arctic grayling, northern pike, walleye, yellow perch, mountain whitefish, burbot, and 10 other fish species (J. Shamess, Impact Assessment Biologist, DFO, Edmonton, AB; pers. comm.).

The company is required to conduct monitoring of compensation effectiveness; to verify fish habitat losses; to participate in ongoing research into the ecological value of end pit lakes, and to support regional multi-stakeholder initiatives designed to address water quality issues, Athabasca River flows, fish health and fish tainting. All monitoring is to be audited by a third party. The obligation to implement monitoring programs is to continue until DFO has determined that objectives of no net loss of productive capacity are met.

Significant monitoring and research initiatives have been, and are being, undertaken in relation to the mining of the oil sands. DFO participates in the Canadian Oil Sands Network for Research and Development (CONRAD) which is a consortium comprising industry, government and universities with a focus on priority environmental research topics.

Placer gold mining

Placer gold mining has been carried out for over a century with little or no effort devoted to the restoration of fish habitat until recently. Accordingly there is a legacy of disrupted and destroyed habitat associated with placer mining.

Primary concerns for the management of fish habitat associated with placer gold mining relate to the effects of elevated levels of sediment deposited in, and suspended in, watercourses and their effects on aquatic organisms, the modification and stability of these watercourses, and attention to habitat compensatory and restorative measures. In many cases existing or abandoned settling ponds at mines frequently fail and therefore provide only short-term storage thus negating any ecological benefit from regulatory requirements that are based on ponded effluent suspended sediment values and dilution requirements.

The effect of sediment on aquatic systems has received world-wide attention for many years. It is a ubiquitous issue particularly because of the potential to impact negatively on aquatic organisms. Sediments are transported into watercourses through many human developments (Waters 1995). Placer miners have contributed to research (e.g. Seakem Group Ltd. 1992) to understand the effects of their activities in the Yukon. There are current and future opportunities to monitor restorative and compensatory measures pertaining to fish habitats.

At present, applications for in-stream mining or stream diversion require a fish habitat restoration plan which is submitted with the *Yukon Waters Act* application to the Yukon Territorial Water Board which, in turn, passes it to interested parties for review. For operations on streams classified as Type I, II, or III, mitigation and plans for compensation and restoration are required.

Under the Yukon Placer Authorization (Government of Canada 1993) it was possible to defer the fish productivity of streams for placer mining. The intent of this provision was for short-term use, but its application resulted in the deferral of water quality requirements of downstream reaches in approximately 80% of licensed operations for streams classified as Type I, II, or III (2003 data, DFO; unpublished information). Without compensation and restoration, the productive capacity of these streams is expected to recover progressively through natural succession events upon the cessation of mining.

There has been no biological evaluation of the recovery of "deferred" streams (1993-2004) following mining. There is a strong concern that natural restoration would not succeed (see Quigley and Harper, and other interviewees' comments in Samis et al. 2005), but this is irrelevant to cases where natural flows are assumed to restore the habitat.

In the northern environment restoration challenges would be greater than in more temperate climes due to, for example, climate and permafrost, that make the successful restoration of streamside vegetation more challenging (S. Gotch, Senior Habitat Biologist, DFO, Whitehorse, YT; pers. comm.). Studies on Yukon streams (2-15-year mining period) show a time-dependent relationship regarding recovery. Also Alaska Department of Fish and Game (1987) concluded that recovery from placer mining would be a protracted process because of the lack of retention of moisture in soils that have had the smaller-sized particles removed (as occurs through sluicing).

Placer operators conducting restoration and compensation in Type IV streams are relying on a guideline that focuses primarily on stream stability and erosion control. It is expected that the restoration of placer-mined streams is carried out in accordance with "Guidelines for the Design and Construction of Stream Channels for Yukon Placer Mined Streams" (Hardy BBT Ltd. 1991). For Type IVa streams the habitat productivity must be restored through channel stabilization and construction of habitat features (rock groups) when mining has been completed. In the case of fish-less (Type IVb) streams the channel must be stabilized when mining is complete. Thus it is expected that there will be recovery of the previously-mined areas and improvements in water quality that will benefit biological resources. However, there have been no evaluations of the efficacy of the YPA requirements for the restoration of Type IV streams. Studies of mines abandoned under earlier regulatory regimes suggest that natural recovery to full productivity is a very long process, and may not occur at all (e.g. Hardy and Associates Ltd. 1981). Whether the current mining circumstances will result in the restoration of productive capacity of streams is unknown.

Stream diversion and compensation: Stream diversion channels to permit in-stream mining and destruction of fish habitat have, typically, been accepted as compensatory structures designed to be stable and withstand flood events (e.g. 1 in 2 years). These measures have generally been on Type IV streams. Scientific evaluations of the success of diversion streams as compensatory structures have not been carried out (S. Gotch, Senior Habitat Biologist, DFO, Whitehorse, YT; pers. comm.).

Reclamation and restorative measures: Placer mining, like other mining activities, generates a number of direct physical effects on the aquatic environment that may include the addition of more sediment and nutrients, and the creation of wider, shallower, and warmer streams, and more wetlands, and secondary channels, or straighter and narrower channels. This is in addition to numerous indirect effects caused by, for example, the accumulation of materials in depositional zones, bed aggradation and increased flooding, greater channel instability and smaller-sized bed material. Fine material results in less obvious morphological changes due to localized deposition, and changes in substrate and water quality for long distances downstream of the deposition. Accordingly, impacts will persist until sediment is evacuated and erosion-resistant riparian vegetation is established. As well, biological impacts will change as channel recovery takes place (Mike Miles, M. Miles and Associates Ltd., Victoria, BC; pers. comm.).

According to Miles, there is much opportunity to carry out remedial work on historically-mined sites, and that recently, mine operators are frequently doing a better job of restoration than that which occurred in the past. That said, he contends that even the best modern reclamation would not meet more southern land-use standards. Mine reclamation needs to be better planned and conducted as mining proceeds. There is a need for more scientific evaluation, training and demonstration projects (Mike Miles, M. Miles and Associates Ltd., Victoria, BC; pers. comm.).

Evaluation and legacy of past mining practices: While there has been an award (Robert E. Leckie) presented to mining operators for "outstanding mining reclamation practices" in recent years, there has yet to be evaluations of the success of the restorative activities regarding the productivity of fish habitat. It is expected that there will be recovery of the previously-mined areas and improvements in water quality that will benefit biological resources. It is likely, however, that decades will be needed before the riparian zone of a previously-mined stream will be fully functional from an ecological perspective. Studies of mines abandoned under earlier regulatory regimes suggest that natural recovery to the previous level of productivity is a very long process, and may not occur at all (refer to Figure 16). Elevated sediment loads may continue for many decades following cessation of placer mining. Lack of channel stability and elimination of habitat, including pools, undercut banks and backwater areas, have been considered to be the greatest limitations to fish habitat recovery in placer-mined streams.

Hardy and Associates Ltd. (1981) suggested that 70 years is required for natural processes to restore stream habitats in wide valleys, but stabilization of streams is over a much longer time frame in narrow valleys. Water quality in wide valleys returned to control levels in approximately 20 years, whereas no predictable trend to recovery was

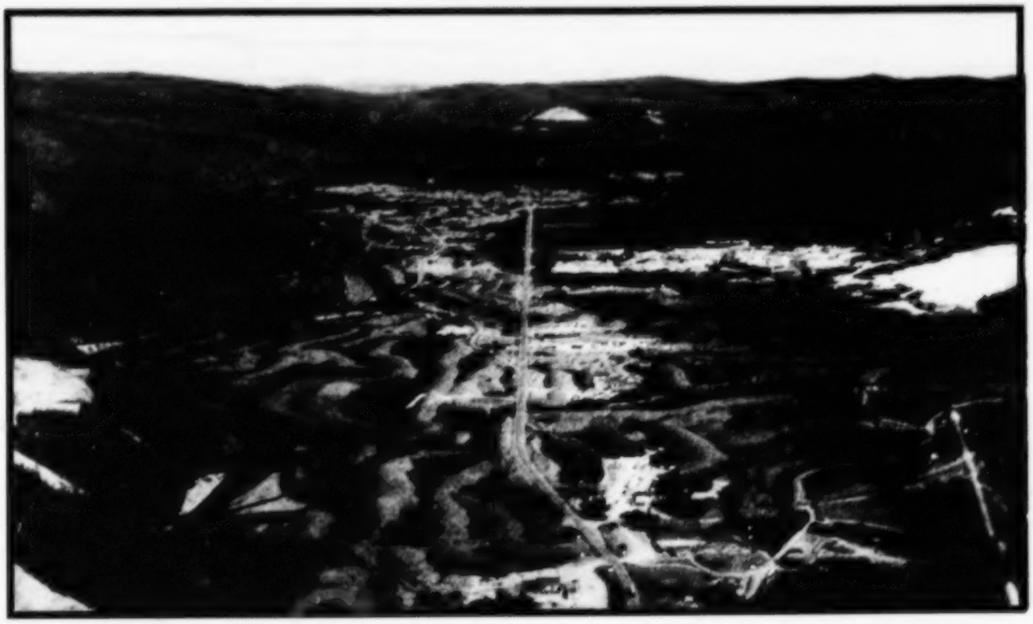


Figure 16. Raised gravel bars along the Klondike River: a legacy of past mining by dredges in the early 1900s.

found in narrow valleys. They assessed the recovery of fish habitat for Arctic grayling in the Yukon and concluded that physical characteristics such as velocity, substrate and channel characteristics were significant factors. Riparian vegetation is a key to stream restoration, and recent studies in the Yukon (Mossop and Bradford 2004) show that 40-80-year-old tree trunks and roots contribute to stream habitat structure. Alaskan studies (Alaska Department of Fish and Game 1987) showed that regeneration of the riparian zone requires the careful redistribution of fine and organic materials in mine spoils, and that vegetation will not grow on tailings that are lacking in organic content and moisture-retention ability. These results mean that at locations where the riparian vegetation has been stripped away, there will be a considerable delay before a properly-functioning riparian zone will be able to contribute to stream habitat structure. Thus, as deduced by Hardy and Associates Ltd. (1981), it is likely that decades will be needed before the riparian zone of a previously-mined stream will be fully functional from an ecological perspective.

Bailey et al. (1998) reported impacts to invertebrate communities in streams that had been mined 10 years earlier. Hardy and Associates Ltd. (1981) found that the densities of benthic communities in wide and narrow valley streams recovered in 5 years following mining. However, the narrow valley streams did not return to control densities and became dominated with sediment-tolerant organisms. Such shifts in benthic invertebrates have been implicated in decreased growth and survival of a salmonid. In this most recent paper, Suttle et al. (2004) reveal the adverse effects of fine sediment within the substrate of a river supporting steelhead trout (see Figure 17). Increasing concentrations of deposited fine sediment decreased growth and survival of juvenile steelhead trout. These declines were associated with a shift in invertebrates toward burrowing taxa unavailable as prey, and with increased steelhead activity and injury at higher levels of fine sediment. They concluded that there was no threshold below which exacerbation of fine-sediment delivery and storage in gravel-embedded rivers will be harmless. However any reduction could produce immediate benefits for "salmonid restoration."

Overall, Hardy and Associates Ltd. (1981) concluded that channel stability and placid flow areas were the most limiting factors associated with the recovery of fish habitat from mining. This conclusion is similar to that deduced by the Alaska Department of Fish and Game (1987) who reported that lack of channel stability and elimination of habitat, including pools, undercut banks and backwater areas, were the greatest limitations to fish habitat recovery.

Elevated sediment loads may continue for many decades following cessation of placer mining as this material is transported to receiving streams. Settling pond failure following cessation of mining operations, or erosion of coarse tailings or stockpiled overburden may also provide substantial quantities of sediment to streams even in the absence of other placer mining activities (Seakem Group Ltd. 1992). Therefore, assumptions that the productive capacity of the affected habitats will be restored by active or passive restoration (depending on the classification), are untested. Available evidence suggests restoration is slow and not assured. Follow-up monitoring must include the establishment of riparian habitat, and criteria are required for monitoring.

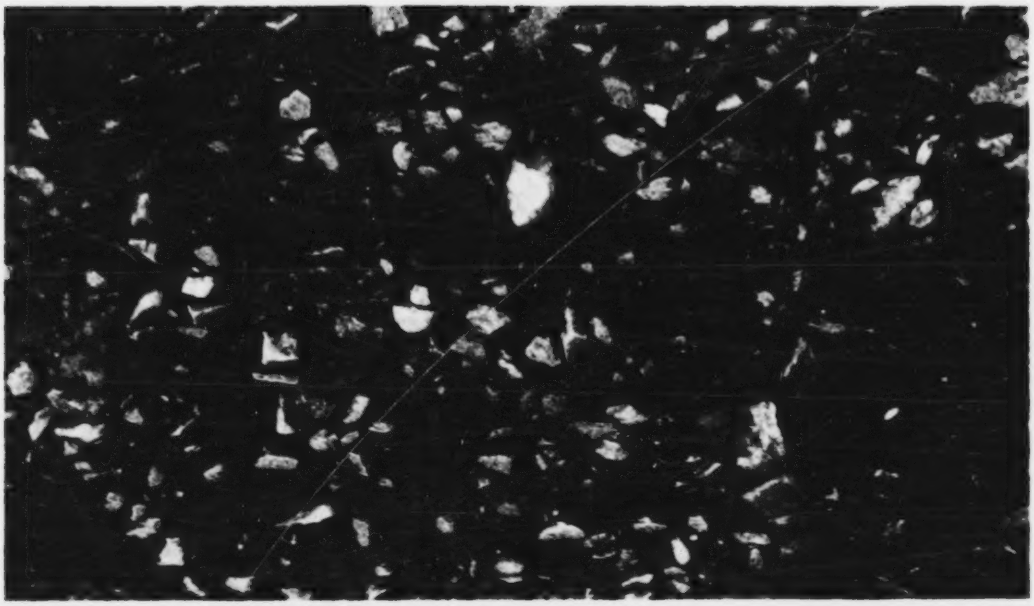


Figure 17. Fine sediment embedded in coarser stream substrate which decreases habitat quality.

Changing regulatory framework: At the time of writing this report in 2004 and 2005, the regulatory and administrative framework regarding placer mining is undergoing change. It is expected that the new regime under which placer mining will operate will be phased in by 2007. Within this new regime is a recognition of the need to improve protection of fish habitat for "Yukon fisheries" and that compensation and restoration are the primary means through which the Habitat Policy goals (*Fisheries Act*) will be achieved (refer to comments from McDougal and Christie in Consultations, Samis et al. 2005).

According to the ISC (2004), scientific information is to be used in the construction of the new regime and as such it will include "approaches to assessing habitat "health"; relation between habitat "health" and fish populations; identification of critical, important and marginal habitat; impacts of sediments on fish and fish habitats; and sediment measuring techniques."

Fish habitat compensation for whole lake and stream destruction

The destruction of lakes for industrial development has escalated rapidly within Canada in recent years. Thirty-one (62%) of the 50 lakes eliminated in whole or parts thereof, occurred over the last 10 years (since 1994), whereas 19 were eliminated in the preceding 5 decades and are still in use or are listed in the MMER (refer to Table 24; it was assumed that government approval was received for historic tailings impoundment areas when they were initially commissioned, hence their inclusion in this table). An additional 20 lakes and a number of streams are under consideration for elimination for mining developments. Thus, 70 lakes or parts of lakes, plus numerous streams, mostly in northern Canada, have been, or are proposed to be, eliminated for use as TIAs (refer to Figures 18 and 19), pit water management basins, or to enable access to ore.

Overall, 70 lakes or parts of lakes, and numerous streams, mostly in northern Canada, have been, or are proposed for, elimination. This is to facilitate the progression of mining activities to use the lakes as tailings impoundment areas (TIAs), waste rock disposal sites, pit water management basins, or to enable access to ore or granular construction materials. Data presented in Table 25 identify the lake, its location and use, and the responsible mining company and sector. The numerous streams that have been, or are planned to be, eliminated or diverted for TIAs and other aspects of mining, including those for oil sands, have not been identified in the table.

Compensatory and restorative measures are required in association with authorizations issued pursuant to the *Fisheries Act* (public documents available at local Fisheries and Oceans offices). Fish habitat compensation agreements have been, or will be, negotiated for most of the lakes that have been destroyed (refer to Appendix, Samis et al. 2005). Exceptions include two lakes that were eliminated prior to the Habitat Policy and two fish-less lakes receiving tailings.

To date, there has been no like-for-like fish habitat compensation for the destruction of a lake.

Figure 18. Diamond mining tailings discharge into dyked-off section of Lac de Gras, NWT.

Figure 19. Aerial view of a tailings impoundment for processed kimberlite at a diamond mine in the Arctic. This portion of the 590-ha lake is receiving, and being filled with, mine tailings; the previous shoreline is visible at the upper parts of the photograph, whereas the extensive deposition of mine tailings has occluded much of the lake in the foreground.

Table 24. The number of lakes approved for elimination/partial destruction in mining operations in Canada, lakes historically used, and those that are under review for elimination before 2009.

Date	Approved by DFO	Historic	*Proposed	Cumulative Number	Cumulative Percent
		15		15	21
1978	1			16	23
1981	1			17	24
1992	1			18	26
1994	1			19	27
1997	12			31	44
2000	10			41	59
2002	2			43	61
2003	7			50	71
Subtotal	35	15	20	70	100
Percentage	50	21	29	100	

* Lakes under review for elimination before 2009

Table 26 provides information on the current status (existing and proposed) of compensation for lake and stream habitat alteration, disruption and destruction, and whether that compensation was monetary, at or away from the mine site, and whether it was like-for-like or unlike in its structure and function. It is apparent that the majority of compensatory measures resulted in unlike habitat (71%), and that was at locations away from the mining site in 65% of 17 cases identified. There was no record of compensation for 19 of the cases identified in Table 26, however most of these mines started operations before the Habitat Policy was developed; financial compensation was accepted for 48% of those compensated for (this included the financial compensation for 5 smaller lakes in lieu of the elimination of the larger Leslie Lake by BHP Minerals under an earlier authorization).

Metal mining discharge regulations and related legislation: Under the former Metal Mining Liquid Effluent Regulations (MMLER, 1977-2002, Government of Canada 1977), TIAs were designated by the Minister. The MMLER were repealed in December 2002 and replaced by the Metal Mining Effluent Regulations (MMER, also made under section 36 of the *Fisheries Act*) which preclude unconfined tailings deposits, and make the inclusion on Schedule 2 of MMER of new TIAs in natural, fish-frequented waters impossible without a Governor in Council (GIC) amendment of the regulations.

Environment Canada (EC) is proposing amendments to MMER. One proposed amendment would make fish habitat compensation a regulatory requirement for future TIAs in natural, fish-frequented water bodies.

Table 25. Lakes listed in MMER as TIAs, other lakes wholly or partially being used for ongoing mining operations, and those undergoing review.

Number	Lake	Company	Approval	Sector	Year	[1]P/T	[2]Use
1.	Wabush	IOC		Metal	[3]H-U	NL	T
2.	Flora	Wabush		Metal	H-U	NL	T
3.	Hessé	Quebec Cartier		Metal	H-U	QC	T
4.	Doré	Copper Rand		Metal	H-U	QC	T
5.	Watson	Noranda (Matagami)		Metal	H-U	QC	T
6.	Petitpas	QIT (Tio Mine)		Metal	H-U	QC	T
7.	Key	Cameco		Metal	H-U	SK	M
8.	Rabbit	Cameco (Rabbit Lake)		Metal	H-U	SK	M
9.	Lower Key	Cameco		Metal	H-U	SK	M
10.	Seahorse	Cameco		Metal	H-U	SK	M
11.	Hourglass	Cameco		Metal	H-U	SK	T
12.	Kathy	Cameco		Metal	H-U	SK	M
13.	Fred	Cameco		Metal	H-U	SK	M
14.	Thompson	INCO		Metal	H-U	MB	M
15.	Misery	INCO		Metal	H-U	MB	T
16.	[4]Anderson	Hudson Bay M & S	MMLER/ MMER	Metal	1978	MB	T
17.	[4]Garrow	Cominco (Polaris)	MMLER/ MMER	Metal	1981	NU	T
18.	East	Claude Res. (Seabee)	FA s. 35(2)	Metal	1992	SK	T
19.	Wollaston (3 zones)	Cameco (Rabbit Lake)	FA s. 35(2)	Metal	1994/ prior	SK	M
20.	Panda	BHP Minerals	FA s. 35(2)	Diamond	1997	NT	M
21.	Misery	BHP Minerals	FA s. 35(2)	Diamond	1997	NT	M
22.	Koala	BHP Minerals	FA s. 35(2)	Diamond	1997	NT	M
23.	Fox 1	BHP Minerals	FA s. 35(2)	Diamond	1997	NT	M
24.	Alexis	BHP Minerals	FA s. 35(2)	Diamond	1997	NT	M
25.	Airstrip	BHP Minerals	FA s. 35(2)	Diamond	1997	NT	M
26.	Long	BHP Minerals	FA s. 35(2)	Diamond	1997	NT	T
27.	Brandy	BHP Minerals	FA s. 35(2)	Diamond	1997	NT	T
28.	Willy	BHP Minerals	FA s. 35(2)	Diamond	1997	NT	T
29.	Nancy	BHP Minerals	FA s. 35(2)	Diamond	1997	NT	T
30.	West Panda	BHP Minerals	FA s. 35(2)	Diamond	1997	NT	T
31.	[4]Albino	Homestake	MMLER/ MMER	Metal	1997	BC	T
32.	[4]Tom McKay	Homestake	MMLER/ MMER	Metal	2000	BC	T
33.	King	BHP Diamonds	FA s. 35(2)	Diamond	2000	NT	T
34.	e1	Diavik	FA s. 35(2)	Diamond	2000	NT	M
35.	e3	Diavik	FA s. 35(2)	Diamond	2000	NT	M
36.	e6	Diavik	FA s. 35(2)	Diamond	2000	NT	M
37.	e7	Diavik	FA s. 35(2)	Diamond	2000	NT	M

Table 25 (cont'd). Lakes listed in MMER as TIAs, other lakes wholly or partially being used for ongoing mining operations, and those undergoing review.

Number	Lake	Company	Approval	Sector	Year	[1]P/T	[2]Use
38.	e8	Diavik	FA s. 35(2)	Diamond	2000	NT	M
39.	e10	Diavik	FA s. 35(2)	Diamond	2000	NT	M
40.	Lac de Gras (part)	Diavik	FA s. 35(2)	Diamond	2000	NT	T
41.	IL1	De Beers (Snap Lake)	FA s. 35(2)	Diamond	2000	NT	T
42.	Triangle	Claude Res. (Seabee)	FA s. 35(2)	Metal	2002	SK	T
43.	Desperation	BHPB	FA s. 35(2)	Diamond	2002	NT	T
44.	Hakim	IOC	FA s. 35(2)	Metal	2003	NL	M
45.	[5]Sable	BHPB	FA s. 35(2)	Diamond	2003	NT	M
46.	[5]Two-Rock	BHPB	FA s. 35(2)	Diamond	2003	NT	T
47.	[5]Beartooth	BHPB	FA s. 35(2)	Diamond	2003	NT	M
48.	[5]Pigeon	BHPB	FA s. 35(2)	Diamond	2003	NT	M
49.	[5]Big Reynolds	BHPB	FA s. 35(2)	Diamond	2003	NT	T
50.	[5]Headwater	Voisey's Bay	FA s. 35(2)	Metal	2003	NL	T
51.	[6]Trout	Aur Resources		Metal	TBD	NL	T
52.	[6]Headwater (on Gill's Brook trib.)	Aur Resources		Metal	TBD	NL	T
53.	[6]Tail	Miramar (Doris North)		Metal	TBD	NU	T
54.	[6]Duncan	Northgate (Kemess N)		Metal	TBD	BC	T
55.	[6]Long	Tahera (Jericho)		Diamond	TBD	NU	T
56.	[6]Unnamed	Tahera (Jericho)		Diamond	TBD	NU	T
57.	[6]Unnamed	Tahera (Jericho)		Diamond	TBD	NU	T
58.	[6]Second Portage	Cumberland Resources		Metal	TBD	NU	T
59.	[6]Winter	Tyhee Development		Metal	TBD	NT	T
60.	[6]Chauve-Souris	McKenzie Bay		Metal	TBD	QC	T
61.	[6]Unnamed	McKenzie Bay		Metal	TBD	QC	T
62.	[6]Unnamed	McKenzie Bay		Metal	TBD	QC	T
63.	[6]Unnamed	McKenzie Bay		Metal	TBD	QC	T
64.	[6]Unnamed	McKenzie Bay		Metal	TBD	QC	M
65.	[6]Unnamed	McKenzie Bay		Metal	TBD	QC	M
66.	[6]Unnamed	McKenzie Bay		Metal	TBD	QC	M
67.	[6]Unnamed	McKenzie Bay		Metal	TBD	QC	M
68.	[6]Laugon	McKenzie Bay		Metal	TBD	QC	M
69.	[6]Coil	McKenzie Bay		Metal	TBD	QC	M
70.	[6]Unnamed	McKenzie Bay		Metal	TBD	QC	M

[1] P/T: province or territory.

[2] Use: T – lake has been, is being used or is proposed to be used as a pit water management facility, a tailings impoundment area, a processed kimberlite containment facility, and/or a waste rock dump.
M – lake eliminated or proposed to be eliminated for other mining purposes, such as for access to ore.

[3] H-U: historic-unknown.

[4] Lakes listed as TIAs in Schedule 2 of the MMER. (In addition, the TIA on South Kemess Creek, BC is listed on Schedule 2, but it is not included in this table of lakes.)

[5] These lakes have been approved by DFO for elimination.

[6] Proposals are under review involving elimination/partial elimination of these lakes.

Table 26. Lakes and streams eliminated or approved for elimination for mining operations in Canada, and authorized compensation for this lost aquatic productive capacity.

AFFECTED WATERS	[1]HADD IMPACT	COMPENSATION					
		Like	Unlike	Offsite	Onsite	[2]DR/DC	$
Wabush	PL						
Flora	L						
Hessé	L						
Doré	L						
Watson	L						
Petitpas	L						
Rabbit	L						
Key	L						
Lower Key	L						
Seahorse	L						
Hourglass	L						
Kathy	L						
Fred	L						
Thompson	L						
Misery	L						
Anderson	L						
Garrow	L						
East	L		X	X			
Wollaston	PL		X		X		
Kemess South	S	X		X			
Panda	PL						X
Misery	L						X
Koala	L						X
Fox 1	L						X
Alexis	L						X
Airstrip	L						X
Long	L						X
Brandy	L						X
Willy	L						X
Nancy	L						X
West Panda	L						X
Streams-Ekati	S	X			X		
Albino	L						
Tom McKay	L						
King	L, S	X			X	X	
e1	L		X	X		X	
e3	L		X	X		X	
e6	L		X	X		X	
e7	L		X	X		X	
e8	L		X	X		X	

Table 26 (cont'd). Lakes and streams eliminated or approved for elimination for mining operations in Canada, and authorized compensation for this lost aquatic productive capacity.

AFFECTED	[1]HADD	COMPENSATION					
WATERS	IMPACT	Like	Unlike	Off-site	On-site	[2]DR/DC	$
e10	L		X	X		X	
Lac de Gras	PL	X			X	X	
IL1	L		X		X		
Triangle	L		X	X			
Desperation	L, S	X			X	X	
Hakim	L, S		X	X			
Sable	L						X
Two-Rock	L						X
Beartooth	L						X
Pigeon	L						X
Big Reynolds	L						X
Headwater	L		X	X			
Total	**52**	**5**	**12**	**11**	**6**	**9**	**16**

L – lake; PL – part of lake; S – stream(s)
[1]HADD – harmful alteration, disruption or destruction of fish habitat
[2]DR/DC – deferred restoration and/or deferred compensation of fish habitat productive capacity

The MMER do not apply to discharges from diamond mines, which are subject to the *Fisheries Act* general prohibition against deleterious deposits into fish-frequented waters (section 36(3)). DFO has authorized lake HADDs from tailings and pit water deposits under section 35(2) and has used s. 52 of the Fishery (General) Regulations to issue scientific fishing licences for fish removal. Using information collected on the species and numbers of fish at the time of "fish-outs" (Fisheries and Oceans Canada 2003) a database is being developed by the Department in collaboration with the University of Alberta to improve predictions of lake productivity to aid decision making.

As stated previously, Environment Canada (EC) has the lead administrative authority for the pollution prevention provisions of the *Fisheries Act,* including s. 36 regulation development, administration and enforcement. DFO's Minister is accountable to Parliament for the whole of the *Fisheries Act,* and is responsible for recommending new or amended regulations developed under that Act to GIC.

Comment

There has been significant effort by industries and regulators to mitigate the impacts of development at a variety of scales, to compensate for effects that cannot be fully mitigated, and to restore fish habitat upon termination of the development. However, in almost all circumstances there is implicit belief that there is sufficient knowledge and understanding of the links between habitat change and consequences to biota, or that there is a minimal or acceptable risk that permits development to proceed (notwithstanding the socio-economic aspects).

The problems associated with the practical application of the habitat provisions of the *Fisheries Act* have been openly reported over many years (e.g. Levings et al. 1997*a*, 1997*b*; Randall et al. 2003; DFO 2004). There has been a progressive but limited response within Science to the deductions of the numerous workshops that have identified the need for more quantitative measures and evaluations of habitat change to facilitate management decisions and improve knowledge: effects that need to be placed in perspective from the site-specific to their role in cumulative impacts over time.

The foregoing comments and examples reveal that there are a variety of methodologies that are used to assess habitat under different circumstances but that all have a common requirement for basic ecological information. In addition, the decisions that have been made regarding compensatory and restorative practices differ among sites across Canada. While such variation in management approaches may well be justified because of site-specific circumstances, the audit trail of decisions and outcomes that is so necessary to scientific and corporate learning (McDaniels and Gregory 2004) is, with only some exceptions, typically rudimentary (e.g. Cudmore-Vokey et al. 2000; Lange et al. 2001; Harper and Quigley 2005*a*; Quigley and Harper in Consultations, Samis et al. 2005). It follows that there is a need for the use of standardized approaches that are based on current and sound science and that there is also a similar need to evaluate the successes and failures of experimental decision making (Minns et al. 1996), particularly regarding habitat change and compensatory and restorative measures. Visual assessments to determine the success of compensatory and restorative actions can have validity (Pratt and Smokorowski 2003) if done scientifically, but this is not likely the case in many circumstances. It is known that fish will use sub-optimal habitats and sometimes to their detriment (e.g. Birtwell and Korstrom 2002*a*). Accordingly, the use of fish presence to indicate suitable habitat or success of compensation is potentially fraught with uncertainty and inaccuracy (refer to Jones and Tonn 2004).

There is a lack of scientific rigor in the general design of projects that evaluate habitat change and especially regarding compensation and restoration (Harper and Quigley 2005*a*; Quigley and Harper in Consultations, Samis et al. 2005). Pess et al. (2003) stated that a major challenge in the restoration and associated planning regarding aquatic ecosystems is the need to make decisions with inadequate knowledge. Inadequate knowledge of system behavior and variability arise from difficulties in defining and understanding complex systems interactions and from not recognizing human limits in controlling natural environments (Pess et al. 2003).

Planning processes have been described in the scientific literature that provide assistance to management decision making related to fish habitat (e.g. Hartman et al. 1996; Pastorek et al. 1997) regarding restoration, and the elements to consider, including information such as genetic diversity (Reisenbichler et al. 2003), the need to use multiple performance indicators to minimize the risk of missing important ecological effects of the project (Pastorek et al. 1997), the critical need to protect the quality and the quantity of habitat, and to define and evaluate associated management actions (Reisenbichler et al. 2003), and the need to study at appropriate scales and to incorporate the required disciplines (Bilby et al. 2003). There is a need to recognize the role of variability and disturbance in maintaining productivity and diversity of stream biota as land use by humans alters the frequency and characteristics of disturbances and tends to homogenize conditions across watersheds rather than introducing diversity. Incorporating this understanding into watershed management and restoration requires that scientists, managers and policy makers view watersheds at much longer temporal and larger spatial scales than is currently the case (Bilby et al. 2003). These authors also stated in the context of river management that "adjusting the perception of regulatory agencies, watershed managers, politicians and the public, that all reaches should conform to some idealized set of optimal conditions at all times will be a major challenge in implementing a more enlightened approach to restoring river ecosystems" Bilby et al. (2003).

Minns et al. (1996) identified the need to focus restoration plans in terms of spatial and temporal scales of ecosystem responses, which are usually greater than the scales of human interventions and assessment. They considered that no single population is the reference point for detecting responses in the fish community. In the same theme, Hartman et al. (1996) stated that it is an oversimplification to attribute population limitation to one factor because fish respond to their total environment. Several physical and biotic factors acting alone or in concert can potentially limit production (Bisson 1989). They shape habitat preference, determine patterns of behavior, and ultimately affect annual production levels. Roni et al. (2002) reviewed watershed restoration and considered that protection of high quality habitats and knowledge of the effectiveness of techniques were important considerations. Pess et al. (2003) considered that watershed restoration that incorporates uncertainty will have the greatest chance of success. Wissmar and Bisson (2003) noted that restoration strategies must consider the resilience of species or the capacity of different fish species to recover from different types of disturbance. However, it is important to identify the limitations of ecological restoration, and plans should include a realistic estimate of the time required to assess whether recovery has occurred. Minns et al. (1996) also indicated that restoration plans should focus more on ecosystems than preferred species. In addition, small increases in supply of habitats critical to a particular life stage, species, guild, or assemblage may not produce changes in the chosen target and "expectations must be approached hierarchically from population, community, trophic level and ecosystem using appropriate measures" (Minns et al. 1996).

Clearly, the issues surrounding the planning and determination of compensatory and restorative measures are complex, but there are some common requirements. As Hartman et al. (1996) stated, programs that are intended to restore natural function to systems or to improve habitat for fish must be planned, evaluated, and reported methodically if they are to succeed and provide information for future programs.

Connectivity and habitat function: Industrial activities can result in the diversion and creation of streams (e.g. in placer gold and diamond mining). The Habitat Policy, through its guiding principle of no net loss, opens the door for such channels to be designed as compensatory structures, and options exist for their use in restorative actions. In both instances, the opportunity to connect water bodies is one of the compensatory options that has the potential for short- and long-term success, and at the same time should not disrupt, but rather complement existing habitat function. Wissmar and Bisson (2003) stated that a greater understanding of the temporal and spatial connectivity of riverine habitats is critical to the survival and persistence of aquatic riparian communities. Losses in connectivity via widespread fragmentation and cumulative degradation of aquatic (e.g. riparian) habitats are major threats to aquatic species and environmental quality. Processes dependent upon connectivity include exchanges with surface and subsurface waters, transport and processing of organic and inorganic matter, and movements of organisms within stream networks and adjacent riparian and upland areas Wissmar and Bisson (2003).

Roni et al. (2002) recommend reconnecting high quality habitats because of the likely quick biological response and the potential for such measures to last many decades. This position is also endorsed by Sedell et al. (1990) who consider that the connecting of key habitats and especially those that act as refuges is a key activity for riverine restoration. Refuges are important as they serve as sources of colonizing organisms and critical seasonal habitats for different life history stages of fish and wildlife, and protective cover and resting areas for migrating species.

Refuge areas also facilitate the exchange of genetic information across generations, and life history diversity in complex habitats (Wissmar and Bisson 2003). Restoring the connectivity among habitats can be essential to allow natural colonization and gene flow from proximate, locally-adapted populations. However barriers to fish movement are desirable when they prevent introgression and hybridization from non-native species (Reisenbichler et al. 2003). Then once connectivity is re-established actions should focus on restoring geologic and riparian processes.

The issues around the use of diversion channels and their success as a compensatory measure is exemplified by the studies of Jones et al. (2003*a*, 2003*b*) and Jones and Tonn 2004) who, in concert with BHP Billiton, researched the function of the Panda Diversion Channel (PDC), a 3.4-km artificial stream, in the Northwest Territories (Figure 20). This research project provides valuable insights into the issue of compensatory streams and their function in the Arctic. The work was carried out in addition to the monitoring of the channel that is being undertaken by Rescan Environmental Services Ltd., on behalf of the mining company (e.g. BHP Billiton Diamonds Inc. 2003, 2004). Collectively, the research and monitoring is providing much-needed information on the biology of Arctic grayling in this region of the Canadian Arctic, and specifically on the success of the PDC as a compensatory stream that replaced braided natural streams in the watershed (see Figure 21). The PDC was constructed to divert drainage waters away from open pits. Unfortunately, no information on the fish use of the streams and resultant production was obtained prior to connectivity being lost between lakes and the streams that were destroyed, nor was there a sound ecological understanding of streams in the Barrenlands (Jones et al. 2003*a*). These are significant limitations to understanding the success of the artificial PDC in compensating for the lost productivity of the natural streams due to mine development. Accordingly, Jones et al. (2003*a*) assessed the effectiveness of the PDC to produce fish against production from reference streams. Once operational, the PDC restored some watershed connectivity allowing fish migration. Arctic grayling were found to migrate through and also use the PDC for spawning in numbers that were 15 times those of the two reference streams together, even though the PDC had 4 times the wetted area (BHP Billiton Diamonds Inc. 2003). It was found that Arctic grayling fry (young of the year, YOY) had an average mass that was 57% lower than that of YOY from natural reference streams. This corresponded to an average 37% difference in standing crop between the artificial and the natural streams for this life cycle stage (Jones et al. 2003*a*; 2003*b*). Temperature did not have a significant effect on the growth; instead reduced amounts of allochthonous and autochthonous organic matter and poor physical habitat appeared to limit the productivity of benthic invertebrates and fish. Jones et al. (2003*a*) speculated that the Arctic grayling might have had to expend more energy obtaining food, thus reducing that which would be available for growth. In addition, the PDC permitted the movement of Arctic grayling and lake trout through it but did not permit the migration of burbot and slimy sculpins.

Jones et al. (2003*a*) concluded that because Arctic grayling have an adfluvial life and migrate among chains of lakes and streams in a watershed, its fragmentation would mean loss of habitat availability and scale, which could have significant population-level consequences (Kentaro and Shoichiro 2002). This comment notwithstanding, it is apparent that, at least over the short term (5 years), the PDC did not function efficiently as a compensatory structure for production of Arctic grayling, and as a migration pathway for all species of fish in the system. Jones and Tonn (2004) commented that the ontogenetic shifts in habitat requirements (simple to more complex with age) could not be met well by the artificial PDC and that, again, this deduction was associated with the poor productive capacity of the channel. Recent unpublished research (D. Janz, Associate Professor, Western College of Veterinary Medicine, University of Saskatchewan; pers. comm.) has further examined the nutritional status of Arctic grayling YOY from the PDC and determined reduced energy stores that strongly infer reduced survival over winter (a topic that has received attention by Lemly 1993, 1996; and Biro et al. 2004).

Based on the movement of fish within the PDC, their continued presence, and the spawning of Arctic grayling, one may assume that this channel was functioning properly as a compensatory structure. However, in that the Habitat Policy relates to the no net loss of the productive capacity of habitats, the findings of Jones et al. (2003*a*) strongly suggest that this has not been accomplished. The results therefore reveal that the end points or criteria for measuring the success of compensatory structures need to be well thought out, and scientifically monitored and researched. The results provide strong inferences that, at least over the short term (years), artificial streams in this region may not contribute viable underyearling individuals to populations of Arctic grayling. The results also emphasize the importance of evaluating the success of fish habitat compensatory projects by using integrated biological, biochemical, behavioral, and physiological metrics at the appropriate life cycle stage. Fish presence is but a coarse indicator of ecosystem health, and if populations of fish are deemed to be the important determinant and integrative measure of successful habitat compensation (Randall and Minns 2000), it is essential that events that detract from the recruitment of individuals to the population are identified and understood. The viability of stream fish habitat

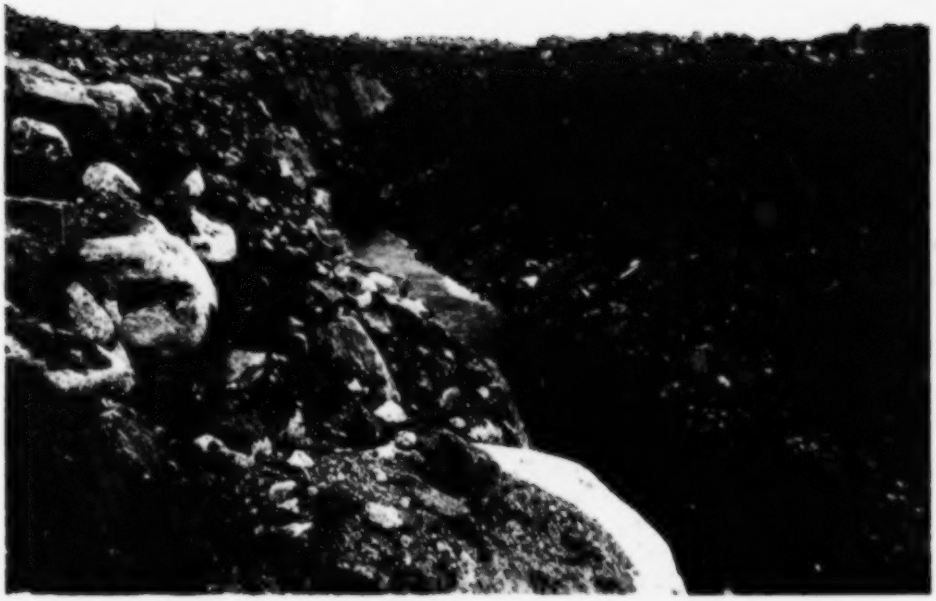


Figure 20. Artificial stream-diversion channel in the Arctic.

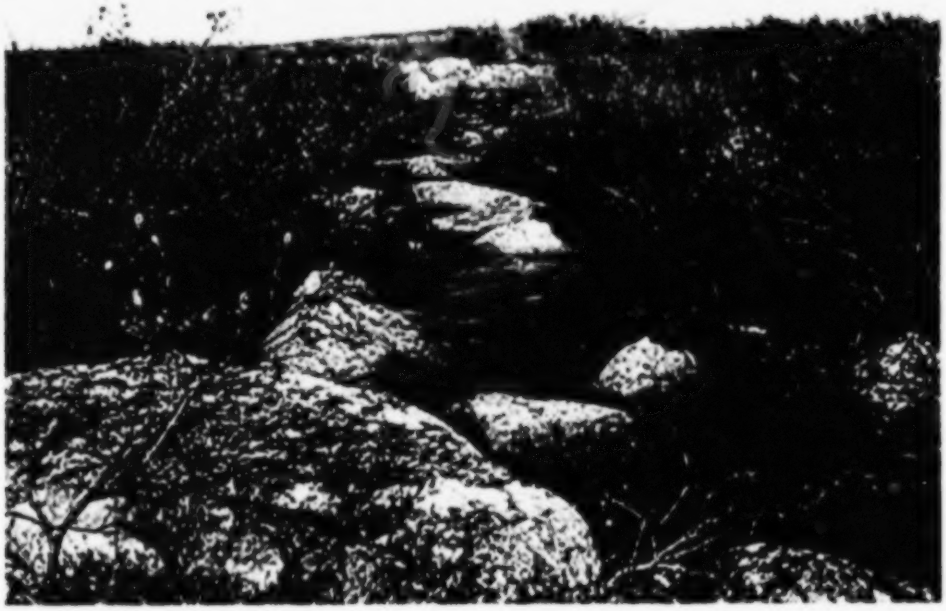


Figure 21. An Arctic tundra stream.

compensatory and restorative projects in ultra-oligotrophic Arctic tundra regions may not be realized for many years if nutrients, and hence food resources for higher trophic levels, remain impoverished. Short-term research on the survival of these fish over winter is feasible and thus it would be possible to determine a criterion of success for habitat compensation measures through the production of individuals with the vitality to enable recruitment to populations of the species.

BHP Billiton provided this opportunity to examine the PDC, and in doing so has provided invaluable information on the likelihood of successful channel compensatory structures in the Arctic. Results of work on the PDC exemplify the need for scientific evaluation and a long-term commitment to fundamental ecological research. The results also provide new knowledge that is of significant practical and scientific importance.

EVALUATION AND AUDIT OF HABITAT COMPENSATION

In 1997 the Auditor General's Report to the House of Commons (Chapter 28: Section 28.58) stated, "In implementing the referral process, Fisheries and Oceans should devote more time and effort to compliance monitoring and follow-up in order to assess the effects of its habitat management decisions and its performance toward the achievement of "no net loss" of habitat" (Government of Canada 1997). Since that time there have been a number of evaluations of the success of the Habitat Policy in maintaining the productivity of fish habitat (e.g. Cudmore-Vokey et al. 2000; Lange et al. 2001; Harper and Quigley 2005*a*, 2005*b*; Pearson et al. 2005; Quigley and Harper in Consultations, Samis et al. 2005) and the net result is not encouraging. The following comments provide for that deduction.

Murphy (2001) reported the results of the application of habitat assessment methodology to changes in lake habitat in Ontario. Non compliance with monitoring requirements was found for the majority of projects sampled and this provided an indication that similar non-compliance issues can be expected for larger projects, where the potential for using a predictive model developed by DFO known as the Habitat Alteration Assessment Tool (HAAT) to determine impacts associated with development applications exists. As such, without the effective use of monitoring strategies, and non compliance with monitoring requirements by project proponents, assurances that compensatory initiatives are satisfying sustainable development objectives are compromised.

Cudmore-Vokey et al. (2000) documented the databases and reviewed habitat compensation and related literature. The literature review was designed to determine if there were any patterns of regionally-recommended documents, document categories, compensation hierarchy, and the proven scientific basis for the use of compensatory techniques. Cudmore-Vokey et al. (2000) found that there was a great difference in the number and type of documents recommended from each DFO region, and suggested that these differences may have been contributory factors leading to differences in decision making, in the hierarchy chosen and the levels of science behind the compensatory activities. They recommended that the database be nationally accessible and maintained to achieve greater consistency and a flow of information, that monitoring compensatory activities, including their effectiveness should be given a higher priority (an integrated framework of monitoring current activities and experimental research would lead to higher-quality management decisions), and that strengthening the scientific basis for habitat compensatory actions would help habitat managers fulfill DFO's commitment to conserving Canada's fish habitat.

Cudmore-Vokey et al. (2000) recognized that more experimental case studies and analyses are required and that learning the effectiveness of compensatory activities should be given a higher priority. The development of standard methodologies for monitoring compensatory programs (including criteria, duration, and success) would encourage consistency, improve decision making and allow flexibility through adaptive management. They suggested that combining research and monitoring would aid management in both the short- and the long-term. They concluded their report, as we have done, by noting that more scientific research is needed on these issues to allow Departmental staff to make more informed, high-quality and consistent decisions with respect to habitat management.

In concert with the review by Cudmore-Vokey et al. (2000), Lange et al. (2001) "synthesized patterns in both the implementation and perception toward habitat compensation" by Departmental staff. A case-study analysis demonstrated substantial variation in frequency of hierarchy level for compensation used among regions and nationwide. The size of compensation ratios (ratio of habitat compensated versus that impacted) differed substantially among, as well as within, regions. In addition, compensation ratios tended to be dramatically reduced in projects with HADD areas greater than 30,000 m^2, which suggested the difficulty in applying the "no net loss" principle in large projects. Lange et al. (2001) also found that compensation success, although limited by the lack of data, varied widely.

Quigley and Harper (in Samis et al. 2005) derived similar conclusions from their recent Canada-wide evaluation of compliance and efficacy of habitat compensatory measures and their evaluation. Assessment of authorizations revealed that in 12% of situations examined there was a net gain in the productivity of the habitat when the average compensation ratio ranged between 4 and 8:1, no net loss in 25% of cases (1:1 ratio), and a net loss in the remaining 63% of cases (0.74:1 ratio). Projects that succeeded in achieving a net gain in habitat productive capacity had compensation ratios of approximately 5:1. Other significant findings and recommendations presented by Quigley and Harper were as follows. The majority of evaluations of habitat compensatory measures were based on determinations of areas of habitat and only one had a scientific research basis (Scruton 1996, cited in Harper and Quigley 2005*a*). Fifty percent of the compensation projects had compensation ratios that were <1:1 ratio (habitat replaced:lost), and 76% <2:1 ratio. The mean duration of post-construction monitoring required by the authorizations was 3.6 years. Quigley and Harper state that there has been little quantitative evaluation of habitat compensation, and there is a need for quantitative scientific methodologies to assess the achievement of NNL through habitat compensation. In most cases, assessments of HADDs and/or compensatory habitats typically included areal measurements of the HADDs and compensation (i.e. no surrogates of productive capacity were sampled). The primary reason for an inability to determine whether NNL was achieved based on a review of the authorization files was due to poor record keeping, a low proponent compliance rate with monitoring requirements, the qualitative nature of the monitoring, and the failure of DFO/proponent to establish a proper baseline in pre-impact monitoring prior to the development (Harper and Quigley 2005*b*). Only a small percentage of the total number of compensatory projects in Canada have been evaluated and the lack of independent, quantitative studies constrains DFO's ability to adaptively manage its habitat conservation program (Harper and Quigley 2005*a*).

They determined that there is a need to select an array of biological indices in order to assess the productive capacity of fish habitat, and a multi-metric approach provides a more complete picture of habitat productivity, rather than simply using fish biomass as an indicator of habitat productivity. Furthermore, in many cases, selecting one surrogate of habitat productivity, rather than an array of ecological indicators at different trophic levels, would have led to erroneous conclusions.

In general, the monitoring requirements for habitat compensatory projects in Canada are inadequate to determine long-term (>50 years) and cumulative ecosystem effects. Temporal losses of habitat productivity are inevitable when compensatory habitats are developed after the HADD occurs. Furthermore, temporal losses are exacerbated due to the time lag until compensatory habitats become functional. In general, compensatory sites were selected opportunistically rather than based on ecological bottlenecks and potential for success.

Limited success in achieving NNL does not erode or invalidate the value of this goal of the Habitat Policy. Rather, it provides the impetus for change. However, some habitats are not possible to compensate for.

Failure to acknowledge the limitations of compensatory science raises the disturbing proposition that Canada's efforts to conserve fish habitat will not be achieving the goal of NNL.

Quigley and Harper deduced that habitat compensation, as currently implemented in Canada, is at best slowing the rate of habitat loss. Improvements in monitoring, enforcement and compensation ratios are necessary. Compensatory habitat should be constructed prior to or concurrent with HADD occurrence and a simple, science-based approach to assessing the effectiveness of compensatory habitat in achieving NNL should be undertaken for most compensatory projects. That is, to quantify the net change in habitat productive capacity, monitoring programs should employ a multi-metric approach (fish density and biomass per unit area, macroinvertebrate density and diversity, periphyton density and diversity), an appropriate experimental design, and surrogate variables for productive capacity measured at HADD, and control sites before and after project implementation. These authors also recommend that monitoring programs should include a pre-impact period of at least 1, and preferably 2 year(s), over a duration of, for example, 10 years which could be "pulsed" (e.g. years 1, 5, and 10). Similar statements have been made by others regarding the need for pre-impact studies and the overall duration of monitoring, both of which relate to the nature of the impact and the environment in which it occurs (e.g. Bradshaw 1996; Lester et al. 1997; Lewis et al. 1996; Jones and Tonn 2004).

Similar conclusions to those identified above were reported by Pess et al. (2003). They stated that a national USA review (NRC 1992) found that many restorative projects failed because they did not consider the broader scales necessary to understand the complexity and multidimensional nature of aquatic ecosystems. Slaney and Martin (1997) agreed that project evaluation is essential to improve program effectiveness, and Reisenbichler et al. (2003) stated that long-term monitoring is essential to provide data for evaluating the progress of restoration programs (e.g. recovery of fish populations) and reducing uncertainties in our knowledge. In 1996 Minns et al. reported some approaches to solutions regarding habitat restoration that included experimentation, science, ecosystem management and coordination.

MANAGEMENT AND POLICY

NEW MANAGEMENT STRATEGIES

DFO, through its Habitat Management Program, has embarked upon operational and organizational changes designed to enable the Department to more effectively carry out its mandate for the conservation of fish habitat in the context of sustainable development (Fisheries and Oceans Canada 2004*b*). This approach is part of an "Environmental Process Modernisation Plan" (EPMP) that contributes to the Government's "Smart Regulation" agenda (Government of Canada 2004*b)*. The 6 elements of the EPMP are a risk management framework, streamlining tools, improving coherence and predictability, renewed emphasis on partnerships, a new management approach to environmental assessment and major projects, and habitat compliance modernization.

Within the EPMP is the requirement for a "Risk Management Framework" (RMF). This framework has its base within science (i.e. the provision of peer-reviewed information). It is to concentrate on the sensitivity of habitat and the severity of impact posing the highest risk to fish habitat, together with an "examination of the public environment and citizens' risk tolerance." The potential effects of a wide range of industrial activities on fish and fish habitat are mapped and presented within Pathways of Effects (POE) models to facilitate this process. Applying approved guidelines and best management practices within the model improves the effectiveness and efficiency of the referral review process for applications for authorizations under section 35(2) of the *Fisheries Act*. It is expected that "the efficiencies of this new approach create obvious benefits to the private sector and to DFO. But even more importantly, it will enable us to do a better job of protecting fish habitat as we target our efforts and resources more strategically."

Another key element of the six-point EPMP plan is the new management approach to environmental assessment and major projects. In recent years the Departmental environmental assessment workload has increased in volume and complexity, and current practices have given rise to various concerns, internal and external, real and perceived, which will be addressed through organizational, policy and governance changes.

The RMF incorporates assessments of a scale of negative effects on fish and fish habitat in relation to sensitivity, thereby providing a means to categorize risk. Thus, the POE approach attempts to standardize the determination of the potential impacts of activities that will affect fish habitat, and to provide this to user groups. Consequently, DFO habitat practitioners, partner agencies and proponents can use POE models to identify potential impacts and concerns and to develop measures to mitigate or avoid effects on fish habitat. Residual effects that cannot be mitigated are then evaluated with respect to the need for compensatory and restorative actions. A risk management framework, or risk matrix, is then used to relate a scale of negative effects and the sensitivity of the fish and fish habitat in order to assign a level of risk to a particular development project. The accuracy with which sensitivity is assigned is a function of the knowledge that indicates its sensitivity. When sensitivity is known the appropriate entry may be made. However, for those areas where this is not known (e.g. northern Canada and ecology of species), caution is required in the use of this scheme. Value judgments play a role within risk management but should be used cautiously. Above all, institutional learning can occur if outcomes are monitored and thereby lead to refinement of decision making criteria (McDaniels and Gregory 2004).

Pathways of effects related to diamond, oil sands and placer gold mining

The Pathways of Effects model is an approach that relies upon quantitative scientific input to identify potential impacts of activities on fish and their habitat and the sensitivity of that habitat and fish to the impacts. Therefore, through its use, and evaluation, information requirements and actions will be identified. A similar approach demonstrating watershed development and effects on communities was shown by Jones et al. (1996). The approach presented by these authors showed how changes in habitat may affect functional linkages (refer to Figure 22). Consequently, their approach provides an example of the linkages between habitat change and ecological effects, and a pathway to determine a rigorous assessment of possible effects. In this example, which was derived from a "littoral centrarchid-urbanization" situation, one of three probable hypotheses of effects is presented. Here, species richness is considered to be the fish community indicator of interest and is the "accepted measure of productive capacity" (Jones et al. 1996). Knowledge existed on the linkage between macrophyte abundance and species richness but in order to understand the cause and effect between them at least 3 linkages must be considered (numbered 11, 12, and 13 in Figure 22). Similarly, mitigation must consider all factors associated with macrophytes for "focussing on only one aspect (say using artificial habitat structure to replace lost cover) would not result in compensation."

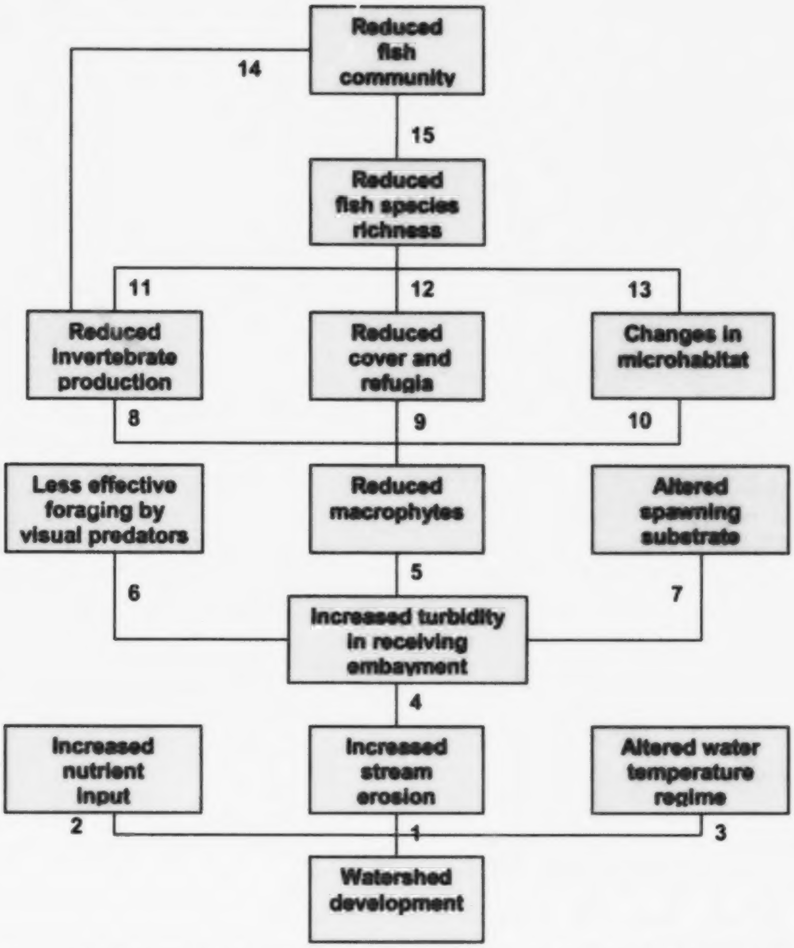

Reduced fish community
14
15
Reduced fish species richness
11
12
13
Reduced invertebrate production
Reduced cover and refugia
Changes in microhabitat
8
9
10
Less effective foraging by visual predators
Reduced macrophytes
Altered spawning substrate
5
6
7
Increased turbidity in receiving embayment
4
Increased nutrient input
Increased stream erosion
Altered water temperature regime
2
1
3
Watershed development

Figure 22. An example of cause-effect pathways from watershed development to fish production (adapted from Jones et al. 1996).

The POE approach was applied as an initial step to the three industrial operations of diamond, oil sands and placer gold mining. The POE diagram that was generated for these three industries is presented in Figure 23. It was considered that there were a number of common issues that could be identified in the exploration, operational and decommissioning phases of these industrial operations that would impact upon fish habitat. By constructing the POE the potential effects of activities are revealed, thus providing an explicit understanding of effects that need to be addressed. That is, effects that can be mitigated if possible, and if not, compensated for together with opportunities for habitat restoration. The POE provides the opportunity to identify areas where the outcome of habitat change is uncertain and where information is lacking for sound decision making. From this follows the opportunity for research to provide the necessary information, decisions that incorporate risk scenarios, and the use of adaptive strategies regarding monitoring and assessment, and overall management flexibility.

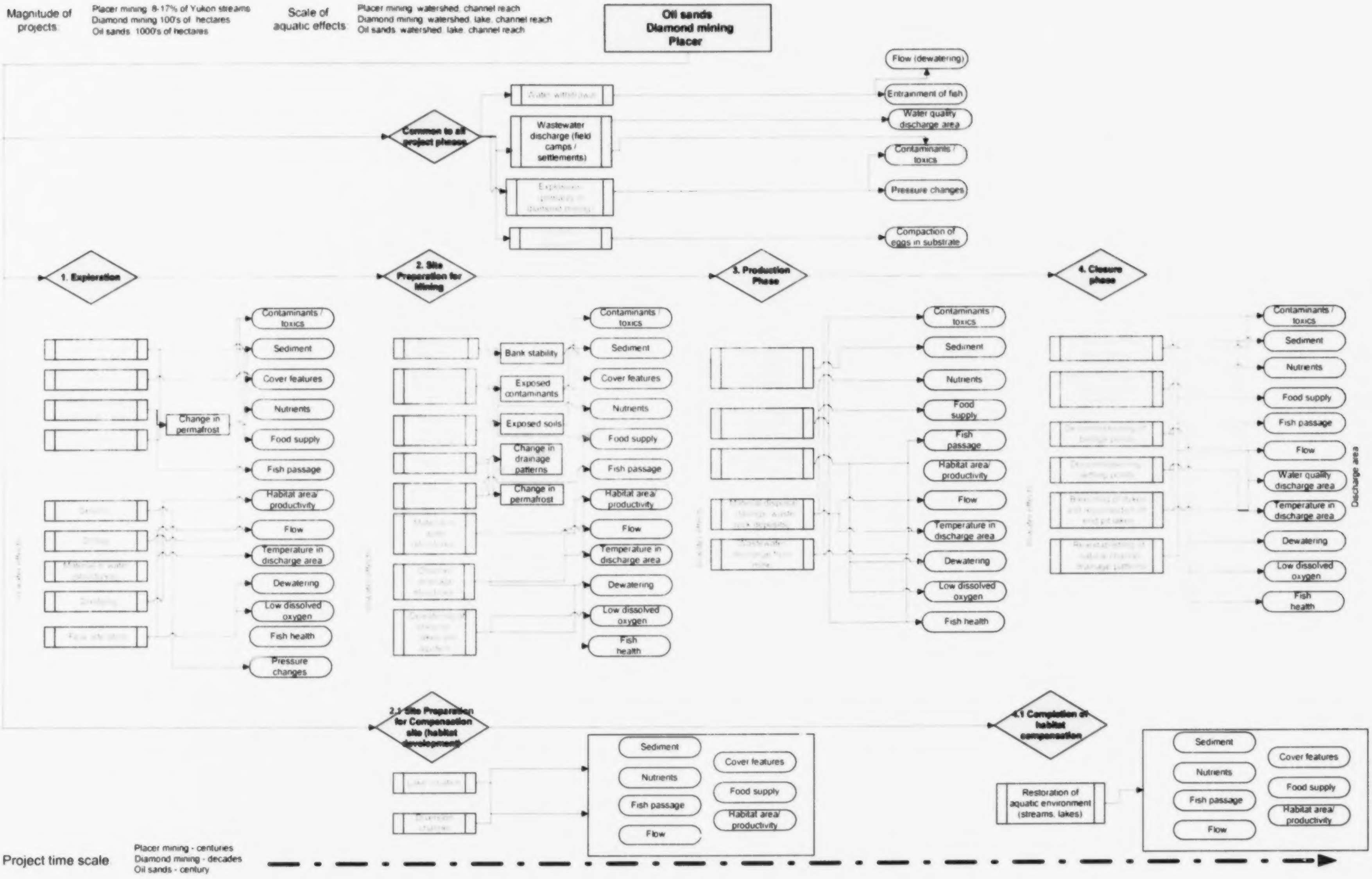

Magnitude of projects:
Placer mining 8-17% of Yukon streams
Diamond mining 100's of hectares
Oil sands 1000's of hectares
Scale of aquatic effects:
Placer mining watershed, channel reach
Diamond mining watershed, lake, channel reach
Oil sands watershed, lake, channel reach
Oil sands
Diamond mining
Placer
Common to all project phases
Water withdrawal
Wastewater discharge (field camps / settlements)
Flow (dewatering)
Entrainment of fish
Water quality discharge area
Contaminants / toxics
Pressure changes
Compaction of eggs in substrate
1. Exploration
Change in permafrost
Contaminants / toxics
Sediment
Cover features
Nutrients
Food supply
Fish passage
Habitat area/ productivity
Flow
Temperature in discharge area
Dewatering
Low dissolved oxygen
Fish health
Pressure changes
2. Site Preparation for Mining
Bank stability
Exposed contaminants
Exposed soils
Change in drainage patterns
Change in permafrost
3. Production Phase
4. Closure phase
Discharge area
2.1 Site Preparation for Compensation site (habitat development)
Sediment
Nutrients
Fish passage
Flow
Cover features
Food supply
Habitat area/ productivity
4.1 Completion of habitat compensation
Restoration of aquatic environment (streams, lakes)
Project time scale
Placer mining - centuries
Diamond mining - decades
Oil sands - century

Figure 23. Pathways of effects diagram relating to the activities of the diamond, oil sands and placer gold mining industries.

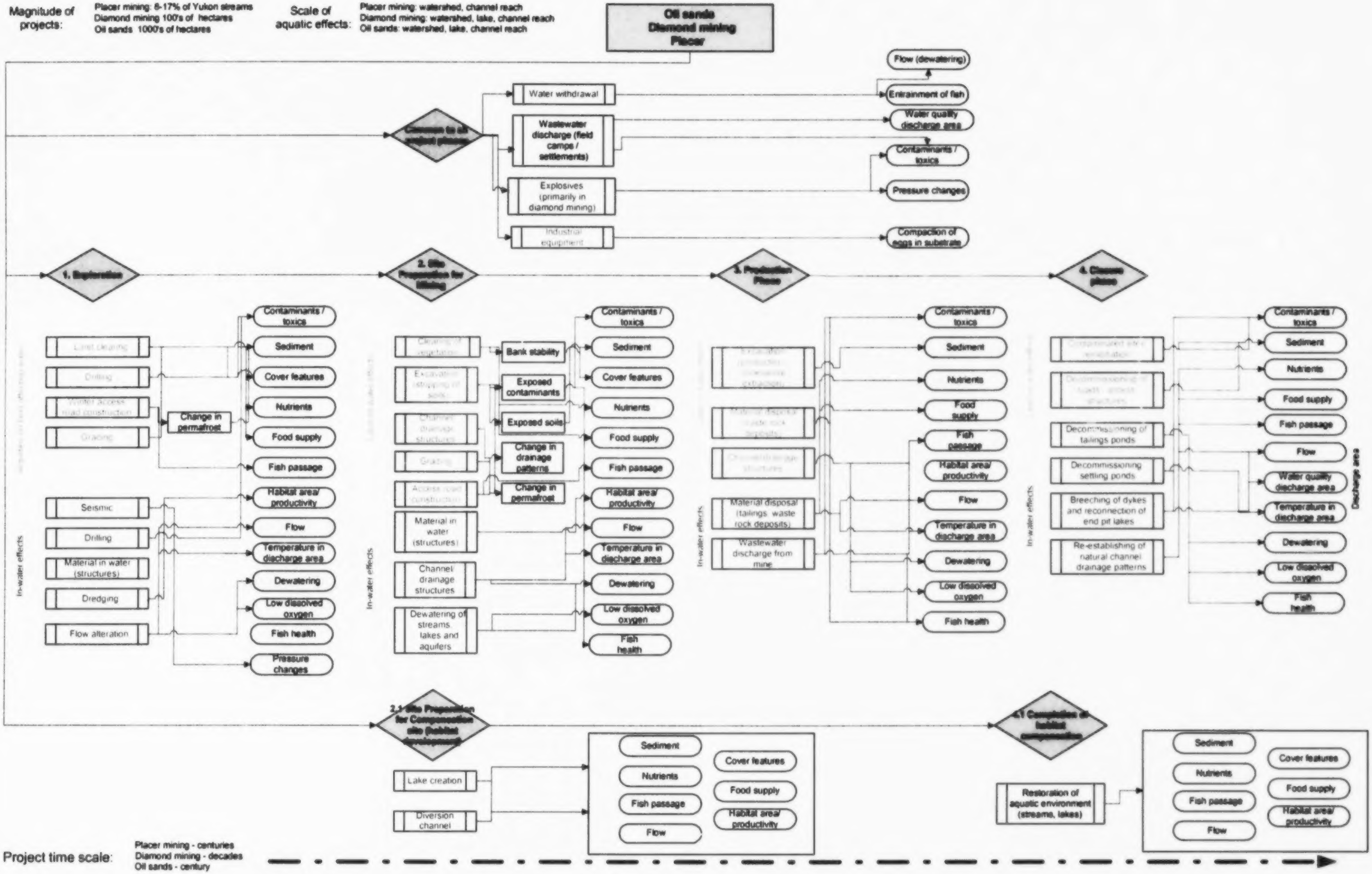

Magnitude of projects:
Placer mining: 6-17% of Yukon streams
Diamond mining 100's of hectares
Oil sands 1000's of hectares
Scale of aquatic effects:
Placer mining: watershed, channel reach
Diamond mining: watershed, lake, channel reach
Oil sands: watershed, lake, channel reach
Oil sands
Diamond mining
Placer
Common to all project phases
Water withdrawal
Wastewater discharge (field camps / settlements)
Explosives (primarily in diamond mining)
Industrial equipment
Flow (dewatering)
Entrainment of fish
Water quality discharge area
Contaminants / toxics
Pressure changes
Compaction of eggs in substrate
1. Exploration
Land clearing
Drilling
Winter access road construction
Grading
Change in permafrost
Seismic
Drilling
Material in water (structures)
Dredging
Flow alteration
In-water effects
Contaminants / toxics
Sediment
Cover features
Nutrients
Food supply
Fish passage
Habitat area/ productivity
Flow
Temperature in discharge area
Dewatering
Low dissolved oxygen
Fish health
Pressure changes
2. Site Preparation for Mining
Clearing of vegetation
Excavation (stripping of soils)
Channel drainage structures
Grading
Access road construction
Material in water (structures)
Channel/ drainage structures
Dewatering of streams, lakes and aquifers
In-water effects
Bank stability
Exposed contaminants
Exposed soils
Change in drainage patterns
Change in permafrost
Contaminants / toxics
Sediment
Cover features
Nutrients
Food supply
Fish passage
Habitat area/ productivity
Flow
Temperature in discharge area
Dewatering
Low dissolved oxygen
Fish health
3. Production Phase
Material disposal (tailings, waste rock deposits)
Wastewater discharge from mine
In-water effects
Contaminants / toxics
Sediment
Nutrients
Food supply
Fish passage
Habitat area/ productivity
Flow
Temperature in discharge area
Dewatering
Low dissolved oxygen
Fish health
4. Closure phase
Decommissioning of tailings ponds
Decommissioning settling ponds
Breeching of dykes and reconnection of end pit lakes
Re-establishing of natural channel drainage patterns
In-water effects
Contaminants / toxics
Sediment
Nutrients
Food supply
Fish passage
Flow
Water quality discharge area
Temperature in discharge area
Dewatering
Low dissolved oxygen
Fish health
Discharge area
2.1 Site Preparation for Compensation site (habitat development)
Lake creation
Diversion channel
Sediment
Nutrients
Fish passage
Flow
Cover features
Food supply
Habitat area/ productivity
4.1 Completion of habitat compensation
Restoration of aquatic environment (streams, lakes)
Sediment
Nutrients
Fish passage
Flow
Cover features
Food supply
Habitat area/ productivity
Project time scale:
Placer mining - centuries
Diamond mining - decades
Oil sands - century

Figure 23. Pathways of effects diagram relating to the activities of the diamond, oil sands and placer gold mining industries.

CONSULTATIONS

Fifty-one people voluntarily contributed to this report through the direct submission of comments and via interviews. Their names, affiliations, and accepted interview comments are contained within the Appendix to this report together with abstracted significant comments organized into 14 key topic areas (Samis et al. 2005).

DIAMOND, PLACER GOLD, AND OIL SANDS MINING INDUSTRIES

Four questions were posed to industrial representatives:

- What are the priority research/information needs regarding diamond mining/placer mining/oil sands development (as appropriate to the specific industry) and aquatic systems?
- What are your opinions regarding collaboration with DFO/universities/other industries?
- Regarding compensation/restoration actions, who decided upon that which was appropriate, who assesses the success/failure of them and what are the measures of success?
- What are the limitations to the growth of the industry, and what is the forecast for the future?

Respondents to these questions were identified by mining sector affiliation as follows: Allison Armstrong, Jane M. Howe, Jayda Robillard, Gord Macdonald, and Robin Johnstone (diamonds); Calvin Duane, Chris Fordham, and Darrell Martindale (oil sands); Mike McDougal and Tara Christie (placer gold).

HABITAT PRACTITIONERS AND MANAGERS

Consultation with this group was considered necessary because of its interface with industry and the need for information to assist with the decisions that must be made regarding fish habitat and developments.

Nine questions were asked in relation to oil sands, placer, and diamond mining, and other industrial operations that have the potential to affect fish habitat (e.g. use of lakes as tailings impoundment areas for metal mining). Representatives from all DFO Regions across Canada were canvassed for their input.

- Is there a formal or informal decision-making process whereby habitat managers are able to assess impacts on fish and their habitat?
- Depending on the answer…what is it or how is it done?
- How are fish and fish habitat loss dealt with in quantitative terms? What are the species/habitats of concern?
- How are cumulative impacts addressed? Is there concern at the fish population level, the lake and/or watershed level?
- What knowledge gaps exist and how is this dealt with in decision making?
- How is uncertainty incorporated into the decision-making process regarding compensation?

- How is habitat restored in the northern locations such as the Yukon, Alberta, Northwest Territories, and Nunavut – as appropriate re industry? What are the pros and cons? Are there case studies (successes and failures)?
- What initiatives (and who is doing them) are underway to fill gaps in knowledge and improve decision making over damage to fish and their habitat?
- What initiatives (and who is doing them) are underway to examine fish habitat compensation and restoration activities?
- Is collaboration with industry on monitoring/research ventures mutually profitable, and are there opportunities to further such co-operation?

Respondents to these questions are identified by DFO Region, as follows: Jeff Johansen and Al von Finster (Pacific Region); Julie Dahl, Jennifer Shamess, Dorthy Majewski, Alan Merkowsky, Derrick Moggy, Tania Gordanier, Ed DeBruyn, and Rich Rudolph (Central and Arctic Region); Sophie Bérubé (Quebec Region); Carole C.J. Grant and Mary B. Dawe (Newfoundland Region).

HABITAT SCIENCE AND NORTHERN ECOLOGY

Discussion with other people focused on selected topics about which they are particularly knowledgeable (Lionel Johnson, Terry Dick, Gordon Hartman, Peter McCart, David Fernet, John Gulley, and François Landry regarding Arctic ecology, Karsten Liber regarding toxicology, Mike Miles regarding geomorphology, Don Toews regarding fish ecology, and DFO scientists and engineers (Colin Levings, Mike Bradford, Martin Bergmann, Susan Doka, Rick Gervais, Chris Katopodis, Marten Koops, Ken Mills, Ken Minns, Michael Papst, Bob Randall, Terence Shortt, Michael A. Turner, and Mike Whittle) regarding fisheries management, ecology, habitat linkages, measurement and validation, and André Isabelle (Natural Sciences and Engineering Research Council of Canada, NSERC) regarding research funds, David Harper and Jason Quigley regarding habitat compensation and assessment, and Paula Pacholek regarding the coordination of northern environmental assessment.

EVALUATION AND PRESENTATION OF COMMENTS AND SCIENTIFIC FINDINGS

The extensive and valuable comments received through these consultations (Samis et al. 2005) were assessed and significant points were combined with information retrieved from the scientific literature. Conclusions and recommendations were drawn from these sets of information, (i.e. from the scientific literature and from consultations). They are presented within the three broad, inter-connected areas of habitat research; mitigation, compensation and restoration; and policy and management.

Common topics from the consultations were searched for within the scientific information base and then combined with those from other consultations to provide un-ranked key topics. As an audit function, the origin of the information used in this step was identified and linked to its source (scientific literature or consultations). Thus the final key topics are those that were identified through the consultations and were also present in the scientific literature that was abstracted for this task.

A list of key topics that combined the comments received via consultations and those in the scientific literature was constructed, initially by using the consultation comments and then seeking matches with information abstracted from the scientific literature. In this way it was expected that more value and importance would accrue to the opinions expressed by those who supplied comments, and the final deduction of key issues. This process included inherent biases as, for example, some of the literature that was assessed was authored by some of the people that were consulted. Similarly, there was the expectation and likelihood that those with whom we consulted also knew some of the relevant scientific literature.

The final 14 topic areas, deduced via the above-mentioned process, represent statements and recommendations that were an amalgamation of comments from scientists, habitat management practitioners, industrial representatives, academics, and other people with an interest in, and knowledge of, the management of Canadian aquatic resources together with findings and deductions from published scientific literature. These deductions are presented below. They provided the basis for the conclusions and recommendations for this report.

1. Biology and ecology of northern and remote aquatic ecosystems

The basic biology and ecology of northern fish species is poorly understood because so little research has been carried out. Specific habitat requirements of certain fish species have not been well documented, and the fundamental ecology of un-impacted lakes and streams in northern Canada is poorly understood. Scientific research that is required to assess ecological effects must be based on understanding the life history of fish, and how they use lakes and streams. Energy flow in nutrient-poor Arctic tundra lakes is not well understood. The factors that control populations of fish in Arctic lakes, and the effects of habitat change, need to be assessed. The paucity of information on certain species will require the incorporation of uncertainty into decision making regarding fish habitat management, and the adoption of a precautionary approach. Government must lead this type of work. Great Bear Lake (the world's most pristine lake) and Great Slave Lake are among the most unstudied lakes in the world.

2. Linkages between fish and their habitats

Although an assessment of the significance of impacts to fish habitat is closely allied to fishery objectives, it has been recognized that fish-habitat linkages are not well known. Habitat assessments should be linked to fish populations so that the impact of habitat alteration is explicitly evaluated at the appropriate geographical scale. This will be possible by provision of long-term data sets on fish productivity and demonstrated linkages with their habitats, an acute issue in northern Canada and especially in the Arctic. The dynamics of fish populations are the ultimate measures of the quality and the quantity of habitat because growth, survival and reproduction are directly related to the requirements for specific abiotic and biotic resources. There is a need for more research on the functional relationships between habitat and the different life stages of fish (habitat-dependent process rates) and productive capacity at the population level. Knowledge of life-stage specific rates and functional linkages with habitat will permit the use of population models to determine critical habitat. The successful restoration of habitat is closely linked to our understanding of ecological processes; if we do not understand the processes, restoration efforts are likely to be unsuccessful.

3. Measurements of productive capacity

Measurements of the productive capacity of rivers and lakes for the purposes of compensation and no net loss assessment are inadequate. There is a paucity of methods to assess habitat loss applicable to fish species in northern Canada. Science-based, simple methods to quantify net change in habitat productivity are needed and a goal should be reliance upon habitat indicators that can be mapped and subsequently validated. Notwithstanding the prior comment, assessing or predicting fish productivity may occur by direct measurement of production rates of all species, or evaluation of biological indices such as biomass and surrogates for habitat variables. However, measurement of productive capacity using biological indices or habitat surrogates is based "on the sometimes untested assumption that these static indices are indicators of the dynamic population processes of recruitment, survival, and growth that together determine production." Validation of the habitat surrogates is needed at the population level, as are methods for quantifying habitat.

4. Program research

The lack of program research is a limiting factor for DFO Science and managers trying to implement policies in northern Canada. This is particularly important because of likely "decades-scale" ecological responses to environmental perturbations. Long-term studies (i.e. 20-30 years) with spatial and temporal controls are necessary and should be commensurate with the longevity of the species. There is a fundamental need for limnological studies and a commitment to long-term research plus shorter-term, related studies that could produce products which have progressive value over time. It is considered that the link between DFO Habitat Management and Science in the north may be lacking, and Arctic science is deficient in DFO. Management and science groups would be assisted by the interaction of monitoring and scientific evaluation of compensation initiatives. New programs in environmental science are needed urgently if the destruction of Canada's freshwater ecosystems is to be prevented. There is a requirement for a long-term commitment to core, freshwater Arctic research. A Network of Centres of Excellence [linked with industry] that deals with people and natural resources and the environment is needed to build "capacity" in the north for decision making. DFO should attempt to set up an LTER tundra network (U.S. Long Term Ecological Research network) which would include other agencies such as Environment Canada (EC) and Natural Resources Canada (NRCan), and universities. The Experimental Lakes Area in Ontario is a model to use in the Arctic to assess the effects of habitat changes.

5. Predictive models and their applicability to northern Canada

Predictive littoral fish-habitat biomass models have been constructed for the Great Lakes, but similar methods have not been developed for northern lakes because empirical data are unavailable. The development of fish-habitat models is lacking for many species, especially those considered to be of "lower value such as forage fish." Most of the development of methodologies has occurred in the more temperate regions and consequently applicability to colder areas such as the Canadian Arctic is, in general, yet to be validated. Some biological information may be anecdotal, and hence DFO needs to address this deficiency and integrate data into models. Quantitative, whole-ecosystem studies of fish and habitat, especially in the development of models and experimental manipulations requires emphasis. The incorporation of habitat into structured ecosystem models is needed. Predictions of fish (species, numbers, biomass, etc.) and their validation within lakes will facilitate decisions regarding the elimination or alteration of those lakes due to industrial development.

6. Databases and archiving

There is no substantive mechanism by which to collect, archive and make available data that have been collected for specific projects and which may be useful to others regionally and nationally. A national archival and retrieval system is required for the management of fish habitat data. Archiving of samples must occur to permit analysis of the collections in the future. Inventories in support of management and research are required. The increased use of inventories can be enhanced (e.g. through use of GIS). A nationally-accessible database which includes background material on specific fish habitat compensation activities and measures of success is required (plus a commitment to maintain it). This will promote consistency and adaptability.

7. Impacts on fish and fish habitat

Little research has been carried out on the loss of habitat and impacts to fish at the lake and watershed level. Cumulative impacts of development and climate change could result in additional stress in the north, but these have not been adequately examined in this region. Clearly-focused research is needed for the development of standardized, transparent, defensible methods to address impacts, to identify best measures of habitat, and the features that are important to fish communities. The evaluation of progress towards quantifying critical habitat and other facets of species recovery programs is an important adaptive management and research strategy. Methods for tracking cumulative change and the interaction of multiple stresses have not emerged. Many Canadian lakes are jeopardized by human intrusion without proper documentation of their baseline state and how humans have altered their communities and biogeochemical cycles. If current trends of fish habitat loss continue, declines in the quality and diversity of freshwater fish resources are certain. Although humans may wish to place their use preferences first, sustained ecological integrity must take precedence if any human-use options are to be retained.

8. Funding and partnerships

A fund needs to be created for research in relation to industrial development in northern Canada. The Natural Sciences and Engineering Research Council of Canada enables academia to obtain NSERC money if there are funding partnerships with industry. Consideration should be given to NSERC grants which would allow for in-kind support from government. These would provide funds for research in the public interest. The needs for research funding and staffing must be assessed in relation to the revenues and/or income from industry such that strong, long-term, ecosystem-based process research should be possible in this fragile, vulnerable and changing part of Canada.

9. Collaborative studies

Projects should be focused and conducted jointly with government agencies and universities. "There are amazing possibilities for joint research" but studies need to benefit companies for this collaboration to occur. Collaboration is an essential part of research and should incorporate traditional knowledge where appropriate. It is the most effective method of achieving results. Opportunities exist for collaborative research on habitat compensation. A consortium that facilitates data sharing is one approach to obtain information that is needed, such as that about fish life history.

10. Fish habitat compensation and restoration

There is a paucity of knowledge regarding lake restoration in northern Canada. Watershed management and restoration requires that scientists, managers and policy makers view watersheds at much longer temporal and larger spatial scales than is currently the case. Successful watershed and habitat restoration requires clear and specific goals, objectives and decision criteria that will allow for accountability and project evaluation. "Appropriate" compensation needs to be defined. Compensatory habitat should be constructed prior to or concurrent with HADD occurrence. Improved access to fish-less lakes and increasing the depth of shallow lakes should be considered as compensation options. A series of experiments could be carried out to assess fertilization effects on dissolved oxygen in lakes over winter. Compensation has typically involved only physical habitat modification, yet there needs to be more assessments of the ability of habitats to augment fish populations by improving their carrying capacity (there is a need to know to what extent fish populations can be increased by habitat manipulation). Holistic studies on stream health and restoration are needed. Effective restoration of river systems has focused on reconnecting key habitats, especially those that function as refuges. Another priority research area is that concerning end pit lakes. Options should be considered to compensate for disturbed habitat with money directed at third party monitoring and research. In addition, compensation should be considered in locations that are more productive than at mines where there is mineralization and low aquatic productivity. Watershed restoration plans need to recognize the role of variability and disturbance in maintaining productivity and diversity of stream biota. Restoration is a poor substitute for habitat protection.

11. Temporal and spatial considerations regarding decision making and monitoring

Temporal losses of habitat productivity are inevitable when compensatory habitats are developed after the HADD occurrence. The losses can be exacerbated during the time for compensatory habitats to become functional. Acceptable time frames are needed for the assessment of productive capacity, time-series monitoring and assessment, and guidelines for compensation monitoring, including the identification of the appropriate scale for measuring compensation and determining its effectiveness. Also necessary is information on habitat dynamics (spatial and temporal variation in habitat use by fishes).

A national review (USA) revealed that many restoration projects failed because they did not consider the broader scales necessary to understand the complexity and multidimensional nature of aquatic ecosystems.

12. Monitoring, assessment and evaluation

There has been little quantitative evaluation of habitat compensation; scientific methods are needed to assess the achievement of no net loss (NNL). Recent evaluation of authorizations revealed that NNL was achieved in 10% of the cases examined. In relation to an assessment of compensation ratios, in 12% of situations examined there was a net gain in the productivity of the habitat when the average compensation ratio ranged between 4 and 8:1, no net loss in 25% of cases (1:1 ratio), and a net loss in the remaining 63% of cases (0.74:1 ratio). Projects that succeeded in achieving a net gain in habitat productive capacity had compensation ratios of approximately 5:1. A simple, science-based approach to assessing the effectiveness of compensatory habitat is required. Projects should be implemented in a tiered fashion that allows information from the results of early tiers to be factored into the implementation of successive tiers. Monitoring the effectiveness of compensation activities should be given a higher priority. Adaptive management requires flexible goals and designs, and a long-term commitment to detailed monitoring and fine tuning after initial implementation. Development of standard methodologies for the monitoring of compensation programs (including criteria, duration and success) is required. Analysis of data is often missing and government agencies should be prepared to take data analysis to another level, and assess it against hypotheses and conclusions. While environmental effects monitoring is mainly focused on chemistry studies of fish, typically assessments are of adults, meanwhile juveniles could be compromised by, for example, food shortages. The provision of data could be a condition of a HADD authorization. Current monitoring requirements in authorizations and licenses are of limited value. Therefore, DFO requires a mechanism and the appropriate expertise to ensure the correctness of experimental design (and replication) so that the results will withstand rigorous scrutiny. It is far easier to protect existing high quality habitat than it is to recreate and restore degraded habitat.

13. Habitat policies and management

Uncertainty is incorporated into a decision-making process regarding HADDs by applying a precautionary approach and professional judgment, but the lack of a fishery is not grounds for devaluing habitat. Direction and a protocol are required from DFO on how to achieve the Habitat Policy's objectives of achieving a net gain in fish habitat productive capacity in pristine areas; a stronger emphasis should be placed on habitat planning. Canada should take a firmer position with diamond mines to ensure progressive restoration as they develop their sites, and fill a previous open pit with waste material from the next pit being mined. If mining for diamonds and the inevitable changes to land and water are accepted (by Aboriginal communities), then perhaps enhancement measures should also be accepted in those areas to offset losses to fish habitat and fish caused by mining developments. Area-based habitat plans were recognized as important in the context of site-specific development activities. A habitat management plan is a plan to conserve, and to meet fisheries management objectives. The analysis of all habitat management issues should begin with a careful articulation of ecological objectives or targets. A risk assessment process for habitat referrals is required.

14. Communication, guidelines and information transfer

A "no net loss procedures manual," "monitoring for success guide" and "standard habitat assessment and protocols guides" are required, together with guidelines to mitigate the effects of exploratory drilling in lakes. There is a need to establish consistent, operational guidelines for determining habitat alteration, disruption and destruction and no net loss of the productive capacity of fish habitat to facilitate consistent decision making.

GENERAL COMMENTS

The comments and information contained in this report were intended to indicate the history and nature of placer gold, oil sands and diamond mining, and the relative effects on fish habitat in northern Canada. Management decisions and options for compensation and restoration of habitats are also addressed in the report. The progression of habitat science is reported to provide information on this aspect of ecological research and its linkage to habitat management. Consultations with a variety of individuals provided valuable information and insight that, when coupled with information obtained from the published literature was used to emphasize significant issues surrounding habitat management and associated science. Specifically, the report addresses deficiencies in knowledge associated with the destruction of habitat in relation to mining and aspects of habitat management including the consistency of application of policies in support of the *Fisheries Act*.

The report serves as a reference source for current (2004-05) circumstances surrounding habitat science and management and selected aspects of the mining sector in Canada. Inevitably, there will be deficiencies in the report because of its broad scope and the gathering together of both opinion and fact. We deliberately chose not to evaluate the comments of others and in that way we hoped to avoid biases that we may have, and not detract from their intent. We used the information that we had gathered to address the issues of information deficiencies and the consistent application of policies regarding the mining industry. In this context we do not impart blame for past events and decisions that have been made regarding habitat management and related scientific research. Rather, we sought to use this historical base to guide future decisions and actions. Through this process conclusions have been drawn and recommendations offered that, if implemented, should assist in more robust decision making through the generation of scientifically-sound information and its accessibility for corporate and scientific learning; benefits from which would accrue to biological resources and their supporting habitats.

INDUSTRIAL DEVELOPMENT

Industrial developments are being facilitated and they are escalating in northern Canada. At the same time they are impacting upon fish and their habitat. With respect to the diamond mining industry, Hanks and Williams (2000) stated that although any given mine is not indefinitely sustainable, given the mineral wealth of the north, the mineral industry is sustainable and that it can provide long-term benefits to the people of the Northwest Territories and the rest of Canada. They also considered that in the absence of regional land-use planning, industry should support a system of well-conceived protected areas that help achieve a balanced pattern of modern land use on the Barrenlands important to make a multiple-use system work in the north. By doing so, these authors suggested that a blended system of land use would limit the potential for the perpetual "nibbling of the environment" that has, in other parts of the world, caused congestion-induced cumulative effects.

It is apparent that these industrial developments are being encouraged and accommodated, but with an expectation that some environmental impacts may be mitigated and, if not, they can be compensated for and habitat eventually restored to permit the sustaining of Canada's natural resources. This expectation falls within our control and "as the dominant species in many ecosystems we are charged with the cardinal responsibility for order and good government within the Laws of Nature" (Johnson 1995). It follows therefore, that consideration must be given to sustaining the productivity of the environment and that "although humans may wish to place their use preferences first, sustained ecological integrity must take precedence if any human use options are to be retained" (Minns et al. 1996). Within this context, the relatively simple and often autonomous Arctic ecosystems (Vanriel and Johnson 1995) provide opportunities for research that will permit a better understanding of increasingly more complex systems (Johnson 2002). Such knowledge and understanding is of critical importance in accurate assessments and predictions of the effects of industrial developments, their potential mitigation, and compensatory and restorative measures.

To facilitate the needs of industrial developments, and at the same time adhere to the intent of the habitat protection and pollution prevention provisions of the *Fisheries Act,* development must be responsible. Ideally it should be with an increased level of certainty regarding the sustaining of the productive habitat base that maintains ecosystems, and their components. Accordingly, decisions that permit development to proceed in regions where the outcome is less certain or more tenuous must be viewed as experimental and evaluated accordingly (Minns et al. 1996). By doing so others may learn and apply the knowledge gained in future decision making (McDaniels and Gregory 2004). This will only be possible through changes to the current knowledge base that is used to assess the implications of habitat alteration and destruction, and through adaptive management. Such a flexible approach would be applied to rigorous, defensible, quantitative evaluations and audits of well-designed compensatory and restorative measures that are designed to meet objectives over relevant scales of time and space (Minns et al. 2001). Inevitably this will require significant scientific rigor and input. Such problems associated with the practical application of the habitat provisions of the *Fisheries Act* have been openly reported over many years (e.g. Levings et al. 1997*a*, 1997*b*; Randall et al. 2003; DFO 2004).

It is apparent from a review of the literature, and the opinion of the scientific and technical communities and industries, that there remains much to do to ensure that the policy provisions for the implementation of the *Fisheries Act* are followed, and that the productive capacity of fish habitat in Canada is sustained. There have been significant efforts by industries and regulators to mitigate the effects of development at a variety of scales of potential impact, to compensate for effects that cannot be fully mitigated and to restore fish habitat upon termination of the development (refer to Samis et al. 2005). However, in almost all circumstances there is implicit belief that there is sufficient knowledge and understanding of the links between habitat change and consequences to biota, or that there is a minimal or acceptable risk that permits development to proceed (notwithstanding the socio-economic aspects). This seemingly-widespread belief perhaps has some foundation in areas where proven compensatory and restorative measures have been applied to circumstances where the knowledge of fish-habitat linkages is better understood, and, therefore, the success of such measures is high (refer to Samis et al. 2005). However, according to reviews of habitat compensatory actions required through *Fisheries Act* authorizations, the measures have generally not met the "no net loss" requirements, and uncertainty of success remains (Cudmore-Vokey et al. 2000; Lange et al. 2001; Harper and Quigley 2005*a*; Quigley and Harper, in Consultations, Samis et al. 2005). Success is, of course, related to the nature of the compensatory measures and the complexity of the problem (refer to Figure 24).

Given the growing prominence of diamond mining and other industrial activities in northern Canada, it is incumbent upon regulatory agencies to ensure the availability of appropriate information in order to make sound and consistent decisions. At the same time, it is also necessary to minimize environmentally-detrimental activities, and make provisions for appropriate compensation and restoration. It is expected that the recent decision by government to employ administrative procedures to make assessments of developments more efficient, such as the use of risk management strategies, and the Pathways of Effects model (Fisheries and Oceans Canada 2004*b*), will assist decisions regarding the impacts of development on fish habitat. However, from a purely practical viewpoint the success of these activities is intimately linked to the information base that provides for the evaluative process. Hence the recognition and incorporation of risk in decision making; the risk being lower the greater the quantity of relevant information used in decision making.

Lakes and streams in the Arctic offer opportunities to examine pristine systems that have been untouched by humans. They have significant value to those requiring ecological information. These relatively autonomous, essentially thermodynamic lakes are simple systems that lend themselves to research, so that not only these systems but more complex ones may be better understood (Vanriel and Johnson 1995; Johnson 2002). As Johnson stated "these relatively simple, autonomous Arctic ecosystems may be regarded as pots of gold at one end of a spectrum of increasingly complex ecosystems." The absence or paucity of information on Arctic ecosystems has become an issue constraining decisions on the mitigation of, and compensation and restoration for, developments that impact fish habitat.

INFORMATION AND ECOSYSTEM COMPLEXITY

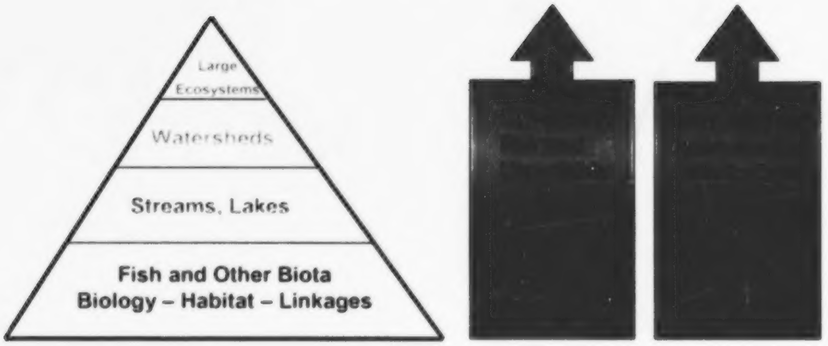
The review of scientific literature related to species of fish in the Canadian Arctic in the region of diamond mining was carried out to assess its adequacy to support decisions that must be made over the acceptability of habitat change and the consequences of those decisions. Research into fish-habitat linkages and direct and indirect (surrogate) measurement of habitat productive capacity have been progressing in more temperate regions because of the better knowledge base that exists in these locations, and the focusing of research effort (e.g. Randall and Minns 2000; Randall 2003; Pratt and Smokorowski 2003). The applicability of most of the methods employed there has yet to be tested and validated under Arctic conditions, notwithstanding the recent research of Jones et al. (2003*a*, 2003*b*, 2003*c*) and Jones and Tonn (2004). Thus sound decision making is jeopardized in such information-deficient regions because of the lack of basic biological and ecological information. Accordingly, risk increases with respect to predicted outcomes of the decisions that are made. Under such circumstances, it is considered that decisions must be viewed as experimental and amenable to experimental design, monitored for their success, and openly reported to aid learning and understanding (Minns et al. 1996; Hartman 2004). The linking of science with management and social objectives and treating each as an experiment will ultimately lead to a greater understanding of how aquatic ecosystems and watersheds function. Strategies need to relate to space and time scales appropriate to definitions of uncertainties, and documentation of resource responses and should be based on science in order to answer questions and formulate testable hypotheses (Collins and Pess 1997, cited in Pess et al. 2003). Rabeni and Sowa (1996) emphasized the importance of having several perspectives from individual fish observation during all seasons to population analysis over a range of spatial scales. At the smallest spatial scale examination of such factors as depth, velocity, and cover, permits determination of those affecting distribution within a reach. Evaluations of populations at the stream-system level allow an understanding of how individual fish preferences relate to amounts of habitat as determined by geomorphic and fluvial dynamic forces and how this influences standing stock throughout a stream system. Evaluation at the eco-regional level shows the overriding influence of physiographic variables on the presence and abundance of fish across stream systems as well as the importance of land use (Rabeni and Sowa 1996).

Various compensatory mechanisms are considered to function within fish populations that mask or otherwise obscure the determination of sublethal effects of environmental change. Due to natural variation and measurement error, several years of monitoring are required to detect large changes to fish populations. Detection of smaller effects requires more years of study and replication (Lester et al. 1997; Walters et al. 1988, cited in Kelso et al. 2001). While there are obvious adaptive capabilities of organisms that allow them to accommodate a wide variety of environmental stresses, the exceeding of thresholds and cumulative stress can both be detrimental to their well being (Wedemeyer et al. 1991). Thus cumulative stress within populations may go undetected because of compensatory mechanisms functioning within the population and the abilities of organisms to adapt, resist or tolerate changing circumstances (habitat) until some threshold is achieved beyond which they show symptoms of debilitation. (e.g. Birtwell and Korstrom 2002*a* regarding the effects of the cumulative degradation of Alberni Inlet, BC, on adult sockeye salmon).

As tools to determine cumulative effects of habitat change on systems (small and large) and populations are imprecise, the additional complication of "natural variability" leads to great uncertainty in predicting and quantifying cause-effect relationships (e.g. Hayes et al. 1996; Bilby et al. 2003; Randall 2003). So it is possible that a system under stress due to habitat change could be perceived as accommodating that change depending on the metric used to derive that conclusion. Frequently the presence of fish is viewed as an indicator of habitat health (e.g. Quigley and Harper in Consultations, Samis et al. 2005). It is a very crude indicator for it is well known that fish will occupy "sub-optimal" habitats and by doing so potentially jeopardize performance, health and survival (Birtwell and Korstrom 2002*a*; Jones and Tonn 2004). Conversely, they may derive an advantage through increased feeding opportunities and protection from predators (Gregory and Levings 1996). Thus, depending on the metrics used, the perceptions of thresholds with respect to populations of fish can lead to erroneous conclusions. The determination of sublethal thresholds with a high degree of resolution to predict population-level responses to habitat change represents a significant scientific challenge. Similarly, the multi-scaled nature of ecosystems and the stress acting upon them present a major theoretical and practical challenge (Lewis et al. 1996). Such comments reinforce the need to address the simplest components of systems in order that more complex systems may be better understood (refer to Johnson 2002).

Kelso et al. (2001) provide a means to assess effects of habitat change using before and after comparisons and this permits lakes to be studied independently and yet still contribute synergistically to a larger experiment. They recognized that extrapolating from the micro to the macro scale is very difficult and whole lake studies costly, but if research groups and agencies act together costs will be minimized to mutual benefit. They recommend carefully-controlled whole lake studies as the best way to provide resource managers with requisite information on managed values such as fish biomass growth and fish community diversity. The Experimental Lakes Area (Schindler 1991, 1995) provides an excellent example of the benefits of large-scale, long-term whole lake studies. They provide highly relevant new information, thereby adding knowledge and a better understanding of the effects of an array of perturbations (e.g. Mills et al. 2002*a*, 2002*b*) that enable habitat and fishery managers to make more informed and hence less risky decisions.

Measures of success and the recovery of fish and habitat from environmental change are, with perhaps some exceptions (e.g. nutrient addition), likely to be a protracted process because of the characteristically-harsh environmental conditions of the Canadian Arctic. The typically low availability of food and cold temperature act together to slow growth rates and influence survival through simple energy pathways (Vanriel and Johnson 1995; Johnson 2002). Furthermore, risk and uncertainty increase depending upon the scale of examination. That is, detecting quantitative changes in large ecosystems is fraught with significantly more inaccuracies in measurement and quantification, than determinations at lower levels of ecological complexity. The success of investigating the biology and ecology of fish is likely a simpler proposition than quantifying and understanding the implications of habitat change within watersheds (refer to Figure 24). Fish populations are viewed as integrative metrics of habitat change (Lewis et al. 1996; Minns et al. 1996; Randall 2003) but resolution of reasons for changes relative to the environment also suffers from inaccuracies inherent in measurement techniques, aside from complicating influences of other variables. Thus detection of change is problematic irrespective of the level of complexity of the biological or ecological entity that is assessed. But it is apparent that increasing ecological complexity compounds the difficulties in quantifying change. Irrespective of these obvious limitations, the basis for sound decision making regarding environmental change lies in understanding fundamental biology and ecology, for they are inexorably linked.



Foundation for sound decision making through knowledge of the interactive functional relationships of systems and their components.

Knowledge base required for assessment of impacts of developments, mitigative, compensatory and restorative measures.

Arctic and northern Canada: ecology of species (habitat requirements, etc.) generally poorly, or not, known. Assumptions create compounded assessment errors, and increase risk.

Figure 24. The basic foundation of knowledge required for sound decision making in relation to ecological complexity, risk, and uncertainty.

ASPECTS OF FISH HABITAT MANAGEMENT (INCLUDING MITIGATION, COMPENSATION AND RESTORATION)

To promote sustainable aquatic ecosystems policy makers can let uncertainty guide the use of the "precautionary principle" in setting limits in human developments and in selecting from a range of restoration strategies, each of which has different levels of risk (Wissmar and Bisson 2003). However, it may not be possible to repair the damage caused by decades of irresponsible land use, for typically we are able to destroy things more readily than we can repair them, thus restoration becomes a poor substitute for habitat protection (Hartman et al. 1996). Although there is a legacy of past developments that have affected fish habitat, current administrative policies (e.g. Habitat Policy), and practices allied to better management of developments are being sought to ameliorate such negative effects. That said, habitat damage is still occurring and that will continue to be the case until it is possible to fully mitigate habitat damage, compensate for residual impacts, and restore that which is degraded. But there might be some regions in Canada where particular developments are likely to cause irreparable damage to fish habitat. The variables that influence the success of habitat compensatory and restorative measures are not only ecological, yet they are of overall importance (Wissmar and Bisson 2003). Without high social commitment, and sound judgment, and the associated economic considerations, the likelihood of success is diminished (refer to Figure 25).

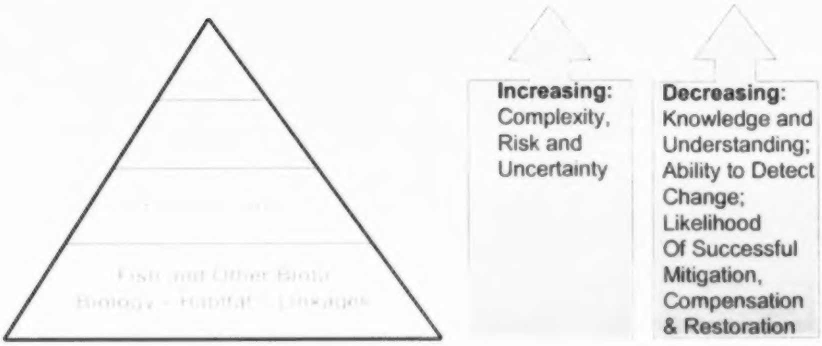

Fish and Other Biota
Biology – Habitat – Linkages
Increasing: Complexity, Risk and Uncertainty
Decreasing: Knowledge and Understanding; Ability to Detect Change; Likelihood Of Successful Mitigation, Compensation & Restoration

Figure 24. The basic foundation of knowledge required for sound decision making in relation to ecological complexity, risk, and uncertainty

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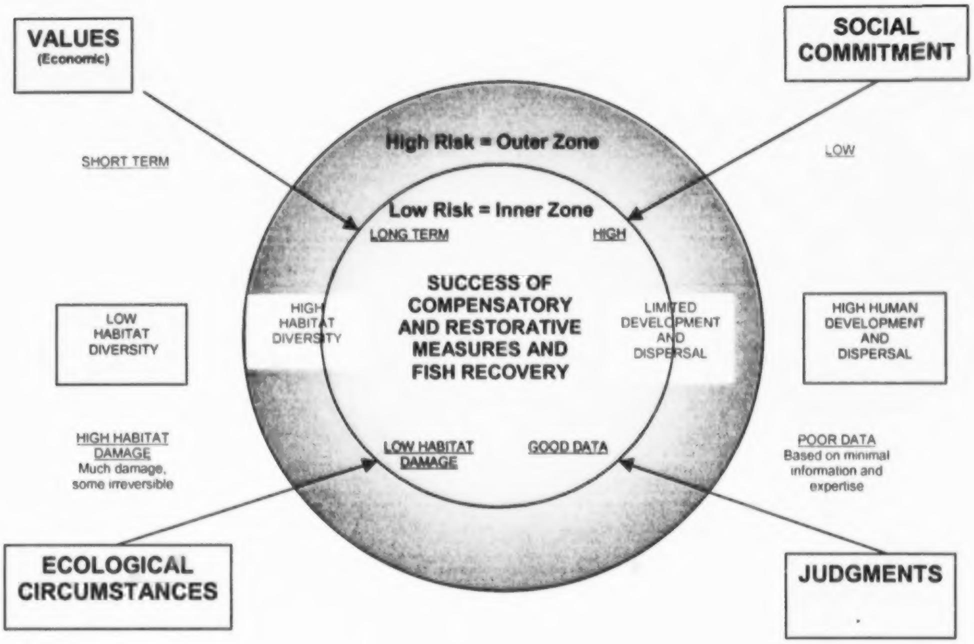
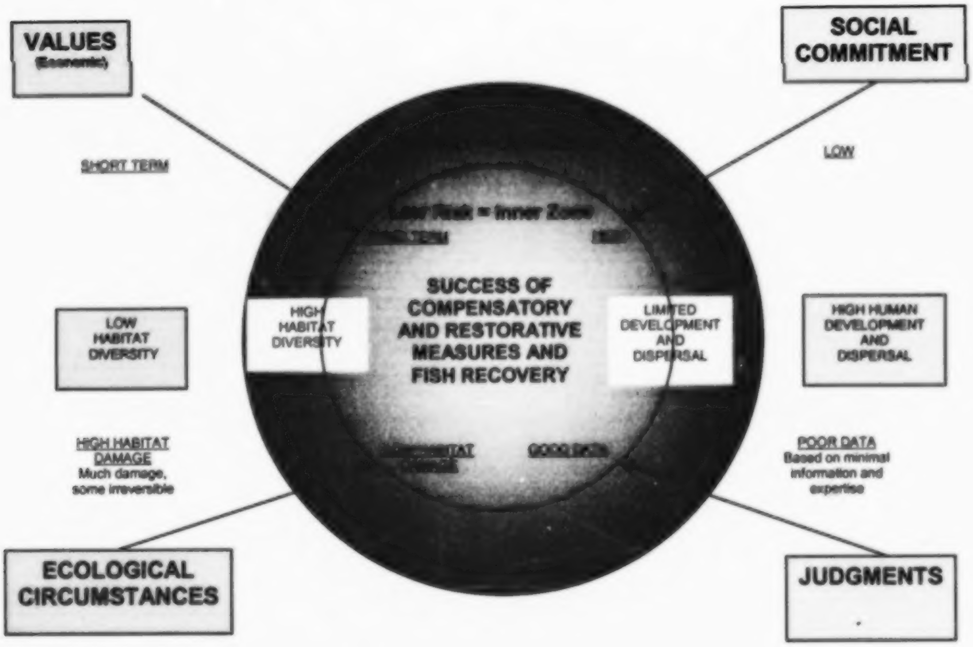

VALUES
(Economic)
SOCIAL COMMITMENT
High Risk = Outer Zone
SHORT TERM
LOW
Low Risk = Inner Zone
LONG TERM
HIGH
SUCCESS OF COMPENSATORY AND RESTORATIVE MEASURES AND FISH RECOVERY
LOW HABITAT DIVERSITY
HIGH HABITAT DIVERSITY
LIMITED DEVELOPMENT AND DISPERSAL
HIGH HUMAN DEVELOPMENT AND DISPERSAL
HIGH HABITAT DAMAGE
Much damage, some irreversible
LOW HABITAT DAMAGE
GOOD DATA
POOR DATA
Based on minimal information and expertise
ECOLOGICAL CIRCUMSTANCES
JUDGMENTS

Figure 25. Diagram showing how the variables of ecological circumstances, social commitment, judgment, and values may influence the success of habitat compensatory and restorative measures (adapted from Wissmar and Bisson 2003).

Knowledge of the simple base of biological and ecological units (Figure 24) permits a better understanding of the consequences of environmental change to more complex systems. However, it is of paramount importance that the assessment of habitat change is made over different scales of ecological complexity and that research and monitoring investigations are carried out over sufficient time frames to permit conclusions to be drawn (Lewis et al. 1996). This is particularly important in addressing the acceptability of habitat change in relation to developments (Jones et al. 2003*a*, 2003*b*; Jones and Tonn 2004). Because space and time together shape ecosystems (Lewis et al. 1996), scale of observation is critical to the design of successful habitat conservation and restoration strategies. Observation over 1 year will lead to a different system description than will observations over several decades; Lewis et al. (1996) suggested that failure to consider the scale of observation is a common limitation to population and habitat studies.

How mitigation takes place and what compensatory and restorative measures occur are a direct function of knowledge about that which is to be changed or lost and then predicting the consequences. Minns et al. (1996) stated that science has been almost absent or neglected when actions to restore ecosystems have been planned. Often science is not used because the objectives of ecosystem restoration are poorly defined. In addition, the


VALUES
(Economic)
SOCIAL COMMITMENT
SHORT TERM
LOW
Low Risk = Inner Zone
LONG TERM
HIGH
SUCCESS OF COMPENSATORY AND RESTORATIVE MEASURES AND FISH RECOVERY
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Habitat restoration strategies are most often developed to help achieve fish population or community objectives. Monitoring programs to assess the success of habitat restoration strategies must be linked to those objectives (Lewis et al. 1996). This most often requires a large-scale study, yet few have set out to determine the temporal and spatial scale required to detect changes in fish abundance and thus test the ultimate success of habitat restoration and conservation strategies.

Habitat protection and restoration should be guided by ecosystem targets and their associated indicators (Kelso and Hartig 1995; White 1991). Once established, managers must be able to measure progress towards them (again, measures must be appropriate in time and scale). Ryder and Kerr (1989) state that habitat needs should be addressed in a hierarchical manner, beginning at the ecosystem level. However, this is difficult to accomplish because of inherent complexities.

A large-scale approach to habitat management will allow for the natural movement of habitat type through space and over time. However, the development of methodologies to detect change at such a level has been slow to emerge.

The results of evaluations of the success in the application of the *Fisheries Act* to prevent habitat loss in the face of development reveal a relatively low level of success (refer to Cudmore-Vokey et al. 2000; Lange et al. 2001; Harper and Quigley 2005*a*; Quigley and Harper, in Consultations, Samis et al. 2005). The reasons for this are numerous, but creating a positive change will require significant effort especially in relation to the setting of clear objectives regarding compensatory and restorative measures and the requirement for adaptive monitoring over time and space that will permit a full evaluation to occur. Such an approach will inevitably lead to better understanding, albeit over time frames that will be related to the rate of habitat change and fish response. Scientific rigor is required in such a scenario, and its importance is greater in those areas where basic and relevant information is lacking.

More information is required to decide upon habitat compensation options in the Canadian Arctic and other remote regions. The Arctic environment has been characterized as being ultra-oligotrophic and as such, while there will be community-based considerations that require incorporation into such plans, success will likely be better if the function of habitat can be retained and enhanced. In that the lakes and streams of the Arctic are typically low in nutrients and are ultra-oligotrophic, fish growth and survival is intimately linked to the provision of food and controlled by relatively-cold waters (refer to Deegan et al. 1997; Buzby and Deegan 2000; Jones et al. 2003*a*, 2003*b*). Thus, as an example, stimulation of productivity can occur through the controlled provision of nutrients leading to enhanced food (Milbrink and Holmgren 1988; Peterson et al. 1993; Slaney and Martin 1997). However, such fertilization requires management to avoid eutrophic conditions (e.g. Rescan Environmental Services Ltd. 2000) that can lead to ecologically-adverse hypoxic and anoxic conditions in lakes, and especially those that are ice covered for most of the year, thereby preventing gas exchange with the atmosphere. Also such changes in nutrient state can lead to ecological shifts such that whereas under oligotrophic conditions primary producers are influenced by "bottom up" control, over time under eutrophic conditions there is a "top down" control (Peterson et al. 1993; Deegan et al. 1997). Thus controlled nutrient addition and management together with improved access to food sources and other important habitat requirements (e.g. habitat complexity) deserve consideration. That is, improving fish access to unconnected lakes that will provide increased feeding opportunities and resources will retain ecological function within a watershed with direct benefits to fish. Such a hypothetical situation is presented within Figure 26. Focused research and monitoring play important roles in decision making and the progression of policies and regulations and their implementation. Such interactions are presented in Figure 27 that serves to encapsulate the prior comments.

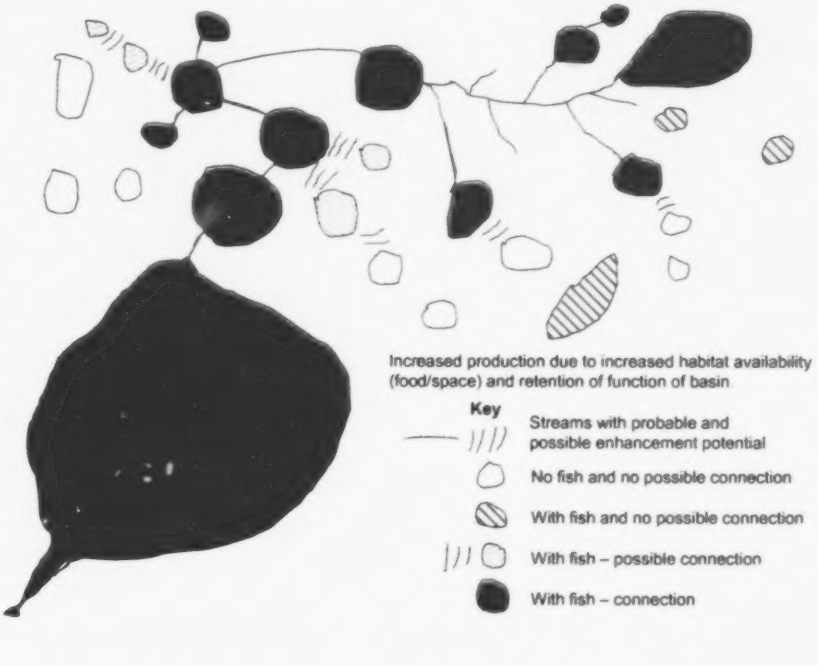

Increased production due to increased habitat availability (food/space) and retention of function of basin
Key
Streams with probable and possible enhancement potential
No fish and no possible connection
With fish and no possible connection
With fish – possible connection
With fish – connection

Figure 26. Hypothetical drainage basin and some connectivity-compensatory options.

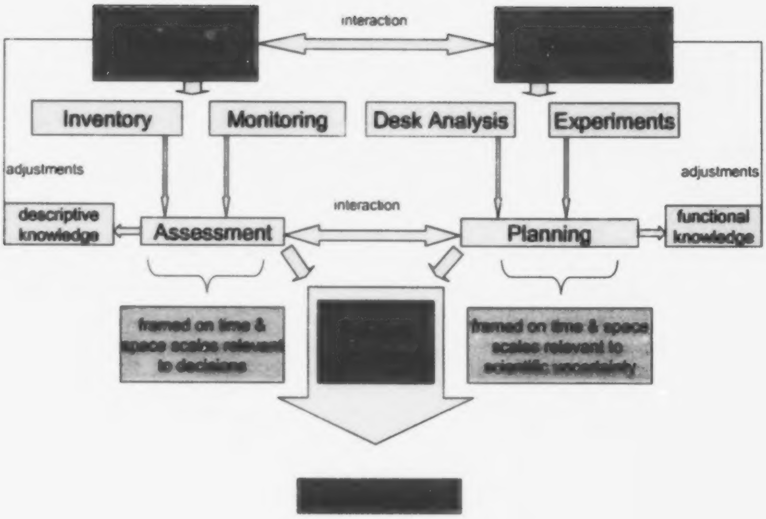
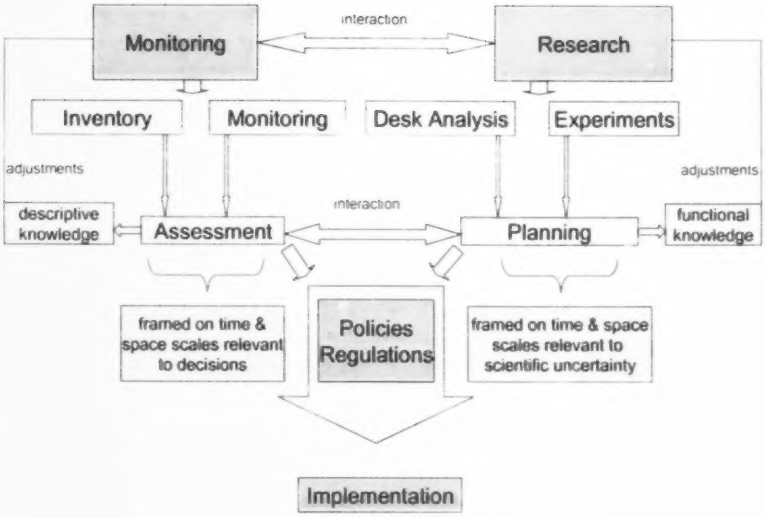

interaction
Inventory
Monitoring
Desk Analysis
Experiments
adjustments
adjustments
interaction
descriptive knowledge
Assessment
Planning
functional knowledge
framed on time & space scales relevant to decisions
framed on time & space scales relevant to scientific uncertainty

Figure 27. Simple integration of research, monitoring and management functions (adapted from Cudmore-Vokey et al. 2000).

POLICIES AND PRACTICAL CONSIDERATIONS

Minns et al. (1996) discuss the issue of ecosystem management, and Grumbine (1994) considered that 5 factors contribute to the long-term goal of maintaining ecological integrity and health. The specific factors are viable populations, representatives of ecosystems, ecological processes, evolutionary potential, and then after these have been attained, the accommodation of human uses. Minns et al. (1996) realized the lower priority given to human use, but considered that sustained ecological integrity must take precedence. Thus the need exists to maintain the productive capacity of habitats through the application of the Habitat Policy (Department of Fisheries and Oceans 1986) in support of the *Fisheries Act*.

Concern continues to exist over the adequacy of information for management of fish habitat in remote and northern regions of Canada (refer to Consultations, Samis et al. 2005). These concerns were addressed by Beecher and Thomas Consulting Services (2003) in their evaluation of the future of aquatic freshwater science in Canada. They recommended the application of an ecosystem approach to research "in Canada's northern areas where ecosystems are especially sensitive to disturbance, and subject to significant modification as a result of climate change." They considered that activities should include the establishment of baseline biodiversity data (as reference benchmarks) for areas that are likely to undergo anthropogenic disturbance, particularly applicable to freshwater ecosystems. They also recommended a commitment to long-term, core, freshwater Arctic research, including areas of expertise such as northern applications of


interaction
Monitoring
Research
Inventory
Monitoring
Desk Analysis
Experiments
adjustments
adjustments
interaction
descriptive knowledge
Assessment
Planning
functional knowledge
framed on time & space scales relevant to decisions
Policies Regulations
framed on time & space scales relevant to scientific uncertainty
Implementation

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Beecher and Thomas Consulting Services (2003) concluded that "given the rapid expansion in development activities expected to occur in the Arctic over the next two decades, it is critical now to begin to develop an expanded scientific capacity to acquire information and provide advice on sustainable development approaches in this less productive and fragile environment. Such capacity is extremely limited at this time, and governments and academia will have to refocus efforts very soon to avert disaster."

Concerns such as those expressed above have also been made by others (e.g. Minns 2001; Schindler 2001). Schindler stated that whereas research on freshwater problems and lakes was once the best in the world, it is now a "national disgrace." While this may at first reading appear to be a radical statement, Schindler supports his statement by reference to the fact that many Canadian lakes are jeopardized by human intrusion without proper documentation of their baseline state and how humans have altered their communities and biogeochemical cycles. Furthermore, Great Bear and Great Slave Lakes are among the most unstudied lakes in the world, and are the only remaining lakes in the world which non-native aquatic species have not invaded. Schindler stated that Great Slave Lake was last studied in detail in the 1950s, and that Great Bear Lake has never been comprehensively studied and currently it is the world's most pristine large lake. Accordingly, there is an urgent need for new programs in environmental science if the destruction of Canada's freshwater ecosystems is to be prevented (Schindler 2001). Minns (2001) cautions that "if current trends of fish habitat loss are maintained in Canada, further declines in the quality and diversity of freshwater fish resources are certain despite our apparent natural wealth." To combat such, modest investments in scientific advice should be made in a progression towards an ecologically-sustainable future (Minns 2001) and, accordingly, much more research needs to be focused on cumulative effects than has been (Schindler 2001).

Fish management (exploited and unexploited resources) and habitat area plans

Fish management and fish habitat area plans are fundamental to the successful management of fishery resources and the attainment of the productivity and conservation goals identified within the Habitat Policy. They represent the starting point for sound management that is stated in the policy, and recognize the need for this integration to guide its implementation. The identification of fish and habitat management goals (such as those for conservation and fisheries) are viewed as requirements that are linked to decisions regarding the acceptability of developments and their impact on fish and their habitat. Minns (1997*b*) describes such needs more fully. Intuitively, one expects that changes in the productivity of habitat upon which fish rely will in some way result in consequences to those organisms. Minns (1997*b*) stated that "in North American freshwaters, most extinction of fish has been due to habitat destruction or alteration," and cited Pearse (1988) who showed that Canadian freshwater fisheries resources were in decline because of habitat loss and over exploitation.

Fish habitat area plans provide the means whereby the goals of the Habitat Policy may be realized. They provide the basis for conserving the productivity of an area. "Effective integration of resource sector objectives, including fisheries, will therefore involve co-operation and consultation with other government agencies and natural resource users" (Department of Fisheries and Oceans 1986). Thus there is an explicit understanding that integrated management will occur. There is also an understanding that this requires an ecosystem approach when one considers that fish productivity is dependent upon the productivity of the system as a whole (Minns 1997*b*). However, rigorous scientific studies are necessary to provide the information that will permit such an understanding to occur.

Habitat management and application of habitat policies and legislation

It is apparent from a review of the literature, and the opinion of people consulted for the purposes of writing this report, that there remains much to do to ensure that the Habitat Policy is not only followed, but that there is a net gain of the productive capacity of fish habitat in Canada.

While science related to fish and their habitats is increasing, little is focused on the links between the production and dynamics of fish populations and communities and the supply and distribution of habitats at various scales (refer to Minns et al. 1996). These authors had the opinion that if current trends of fish habitat loss continue, declines in the quality and diversity of freshwater fish resources are certain. Part of the reasoning is associated with the opinion that habitat management is still mainly reactive and that methods for tracking cumulative change and the interaction of multiple stresses have not emerged. They considered that management needed to take a more proactive approach supported by better deployment of available science and scientific methods that should emphasize quantitative, whole-ecosystem studies of fish and habitat, especially in the development of models and experimental manipulations.

Inconsistency in application of the *Fisheries Act* among mining sectors

Differences exist in the application of the pollution prevention and habitat protection provisions of the *Fisheries Act* among the mining sectors identified in this document. However, there is recognition in government of the need to address this issue.

Policy and regulatory issues: DFO has been inconsistent in implementing its policy and regulatory responsibilities in the context of mining in Canada's north. In addition, it appears there has been an overall net loss of fish habitat productive capacity due to human developments. In this regard, we have chosen to avoid commenting on the responsibility for past events and decisions that have been made regarding habitat management and related scientific research. Instead, we have endeavored to use this historical base, consultations, and information from pertinent literature, from which to move forward by recommending measures to facilitate habitat management and related science. By doing so there should be a benefit to aquatic resources, and also mutual benefits to industry and regulators.

The discharge of sediment into waters frequented by fish often occurs due to mining. Minimizing or avoiding its input into such waters has been recognized as an important step in the maintenance of most fish habitat (Waters 1995). Placer mining, by its very nature, has the potential to add large quantities of sediment into watercourses, and attention has been given in this document to industry and habitat management to reveal some of these issues around human developments that have occurred over many years in northern Canada.

Placer mining has occurred in the Yukon for over a century, typically by small operators on very limited budgets. Sediment discharges are regulated under the *Fisheries Act* using a section 35 class authorization (Government of Canada 1993) which allows the harmful alteration, disruption or destruction (HADD) of fish habitat based on stream classification, which is a function of fish use. For example, for those streams considered more valuable, proposed habitat disruptions must be compensated for before mining commences. For other streams considered less valuable, habitat must be restored, or the channel stabilized, when mining is complete. Placer operators reportedly complete a worksheet on gradient and width, but it appears the work is inadequately monitored, and it is subject to disturbance from the effects of mining upstream.

To date restorative actions related to placer mining have been few, but in compliance with a new regulatory regime (as noted elsewhere in this report), "compensation and restoration are the primary means through which the policy goals (*Fisheries Act*) will be achieved where there are short-term disruptions of habitat" (ISC 2004). The net damage to fish habitat from placer mining may be reversible in some areas, but over a long time frame (Hardy and Associates Ltd. 1981; Alaska Department of Fish and Game 1987).

Diamond and oil sands mining, in contrast to placer mining, are conducted by comparatively well-capitalized interests, and the discharges are managed under stringent territorial or provincial water licences.

Fish habitat authorizations for diamond mining have been used to allow HADDs caused by lake draining, stream diversions, and the use of lakes to manage pit water, and as depositional areas for waste rock or tailings. DFO has required compensation to offset damage to fish habitat from whole lake destruction in diamond mining; however, successful compensation has not yet been documented. This is in part due to the lack of proven compensatory measures for northern habitats, and because of the paucity of research that has been conducted on fish-habitat requirements in the north. Furthermore, there is community resistance to enhancement or modification of pristine fish habitats, thus presenting a limitation to compensatory measures that could be employed (J. Dahl, Area Chief, Habitat Management, DFO, Yellowknife, NWT; pers. comm.). Notwithstanding these comments there is, however, community support for the economic benefits that diamond mining provides, despite the fact that it often has resulted in whole lake destruction.

Diamond mining companies excavate multiple open pits concurrently, without being required to progressively restore mined-out pits in succession. As a result, achievement of no net loss of the productive capacity of fish habitat is being deferred, and consequently the development of end-pit restoration technology is hindered.

The Habitat Policy's hierarchy of preferences requires assessment of like-for-like compensation before consideration of moving offsite with replacement habitat

(Department of Fisheries and Oceans 1986). There have been many lakes eliminated for diamond mining developments, however, no new lakes have been developed as replacement habitat at these mines. It appears that damage to fish habitat in the tundra due to diamond mines is likely to be largely irreparable, in that end pit lakes or tailings impoundment areas are unlikely to be fully restored. A few compensatory measures that offset fish habitat damage at diamond mining operations in the tundra are being evaluated scientifically (e.g. Jones et al. 2003*a*, 2003*b*; Jones and Tonn 2004), and are also being monitored for success (BHP Billiton Diamonds Inc. 2004) – this will take many years.

In the oil sands industry, large-scale surface disruption occurs, and this can include stream destruction approved by DFO using a section 35 authorization. (Jennifer Shamess, Impact Assessment Biologist, DFO, Edmonton, AB supplied the following comments.)

DFO reviews each development application to understand the implications to fish habitat, both locally, and in the lower Athabasca River watershed. In addition to authorizing the construction of ecologically-functioning watercourses as compensation, DFO has agreed to allow developers to also construct compensation habitat that is unlike the natural habitat that is damaged (i.e., lacustrine for riverine). The rationale for this is two-fold: there are limited sites within any particular oil sands lease area that are suitable for construction of a permanent waterbody, and, stakeholders have consistently insisted that compensation structures must be constructed and operated close to their communities. Pit lakes (contain tailings) have not yet proved capable of sustaining healthy fish populations and so are not considered compensation habitat. These lakes may eventually provide suitable habitat but that deduction will probably be at least 50 years from now (the temporal destruction of habitat and its replacement is taken into consideration when DFO when evaluates these projects). The lakes that are accepted are designed to provide habitat for specific fish species and will be monitored closely to evaluate their success (J. Shamess, Impact Assessment Biologist, DFO, Edmonton, AB; pers. comm.).

The effectiveness of replacing streams with diversion channels and artificial lakes is, however, untested. Residual hydrocarbon contamination can occur in surface waters in oil sands development areas. Research is underway through the Canadian Oil Sands Network for Research and Development (CONRAD), a consortium of companies, universities and government agencies organized to facilitate collaborative research in oil sands, including environmental research.

Differences exist on a mining sector-wide regulatory basis between metal mining and diamond mining in the context of tailings impoundment area designation. Regarding the former, natural, fish-frequented water bodies are scheduled as tailings impoundment areas (TIAs) through a GIC amendment to MMER (Government of Canada 2002). Whereas, TIAs in diamond mining are authorized by DFO officials using the section 35 HADD provisions of the *Fisheries Act*.

The acceptance of money as partial compensation for lost and degraded habitat has occurred in relation to the development of a diamond mine in the Arctic. The money was placed into a federally-managed Habitat Compensation Fund. The acceptance of compensatory funds is a variation to standard practices that seek options to employ fish habitat compensatory and restorative measures to offset such loss and degradation (Fisheries and Oceans Canada 1998). The unique Arctic circumstances together with a general lack of ecological information and non-validated habitat assessment, compensatory and restorative measures, and limited options, no doubt contributed to the acceptability of this choice.

The application of the Habitat Policy differs among the Department's regions across Canada, and in relation to many understandable factors. However, for transparent, rigorous and consistent decision making that is congruent with the new initiatives being undertaken by government, (Fisheries and Oceans Canada 2004*b*) it will be necessary for staff to have better guidance and supporting information. The choice and use of appropriate measures to assess the consequences of habitat change and thereby help to mitigate loss and compensate and restore habitat, will be dictated by site-specific circumstances. However, through the provision of appropriate guidance at the national level a more consistent approach will occur and learning will be facilitated. Directives for staff to promote rigorous and consistent decision making require a good science base, which is lacking, particularly for northern fish and their habitats.

These comments are not to be misconstrued, for significant and valuable effort has already gone into the provision of such approaches (e.g. with respect to the application of the Habitat Policy, and use of scientific methods). However, there are constraints that can only be offset through the provision of new information if greater consistency and less risk in decisions are required. This is particularly true for remote and northern parts of Canada where our understanding of ecology is very limited, and where validation of the existing assessment and evaluation methodologies has yet to occur.

With respect to placer mining, it will be necessary to place a high priority on the implementation of habitat compensation activities and the monitoring of them and restorative works under the new regulatory regime in order to move toward the achievement of no net loss as set out in the Habitat Policy (Department of Fisheries and Oceans 1986). Regarding diamond mining, it will be important to ensure that no net loss measures, including those involving enhancement of pristine habitats, are accepted by affected communities as an integral part of the mine development project. Diamond mining companies should be required to progressively restore mined-out pits to expedite the development of successful restoration techniques, and the achievement of no net loss. Given the paucity of research on northern fish species and their critical habitat requirements, DFO should be developing a policy framework to delineate conditions under which money could be acceptable compensation for whole lake destruction. Presumably this would assist in funding key research issues to address information deficiencies that would promote better understanding and decision making of mutual benefit to industry and regulators, and the sustaining of aquatic resources.

In the case of projects requiring whole lake destruction, rather than being constrained by the Habitat Policy's preference for like-for-like compensation, DFO should be encouraging proponents to explore compensation options. For instance, options that support fish management plans and/or local area plans, and in so doing, the focus should shift to compensation aimed at eliminating habitat "bottlenecks" to aquatic productivity.

Following the successful model of CONRAD, DFO should champion the establishment of consortium-based research with the diamond mining industry, universities and government agencies, in order to facilitate collaborative research into fish habitat compensation and restoration in the north.

The approach of developing *Fisheries Act* section 36 regulations for diamond mining discharges and their associated tailings impoundment areas should be discussed between key federal departments.

Hypothetical management plan

A simple plan is provided based on the foregoing comments in this report. It is presented in Figure 28. It is not all inclusive but serves to illustrate the components of the assessment of development that will impact fish habitat and the link to meet the overall goals of the Department. In this hypothetical situation it is assumed that a productive lake will be destroyed and that there will be compliance with the habitat protection and pollution prevention provisions of the *Fisheries Act*.

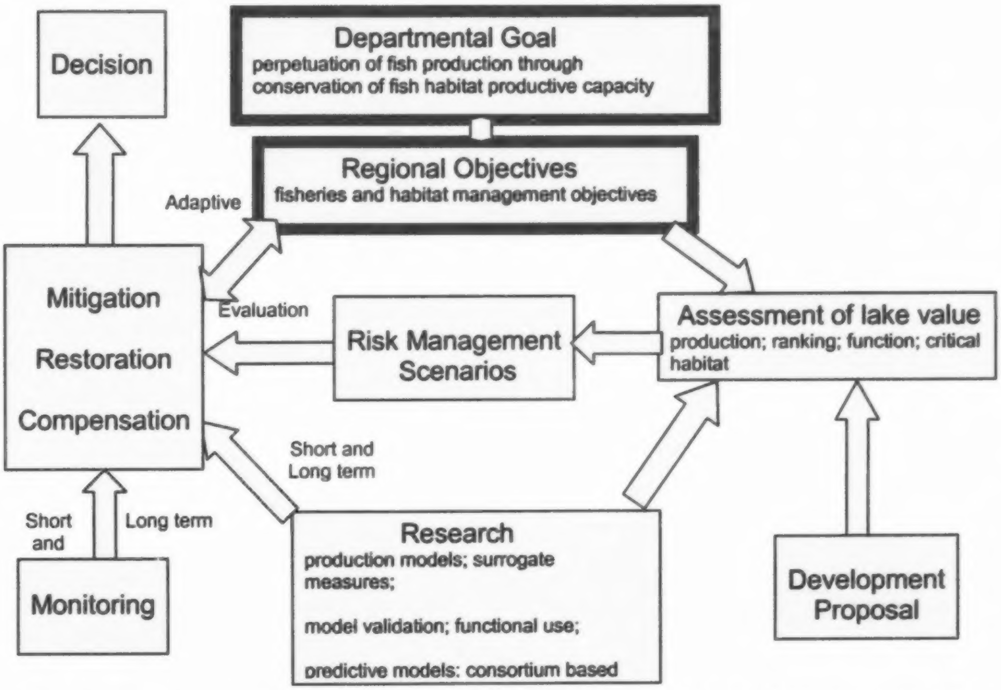
Decision
Departmental Goal
perpetuation of fish production through conservation of fish habitat productive capacity
Regional Objectives
fisheries and habitat management objectives
Adaptive
Evaluation
Mitigation
Restoration
Compensation
Risk Management Scenarios
Assessment of lake value
production; ranking; function; critical habitat
Short and Long term
Short and
Long term
Monitoring
Research
production models; surrogate measures;
model validation; functional use;
predictive models: consortium based
Development Proposal

Figure 28. Processes and linkages regarding the assessment of a hypothetical development proposal that will eliminate a productive lake, and the relationship to Fisheries and Oceans' regional and Departmental objectives and goals.

Regional fish production, harvesting and conservation objectives (including genetic considerations) are the fundamental starting points that should guide decisions on the acceptability of habitat change (e.g. Jones et al. 1996; Lester et al. 1997; Minns et al. 2001). Without such targets at various scales of magnitude, progress cannot be assessed, nor can the steps taken.

It is accepted that the setting of management objectives in some regions may be difficult, but they are fundamentally important. Thus, by setting such objectives the risk of loss of a productive lake may be assessed in an appropriate reference framework linked to fish management objectives (refer to Figures 28 and 29).

At this assessment stage the initial steps could include the identification of potential and predicted lake productivity by employing standard measures, then assessing the loss of this lake in relation to the ranked importance of lakes within the watershed and larger ecological unit.

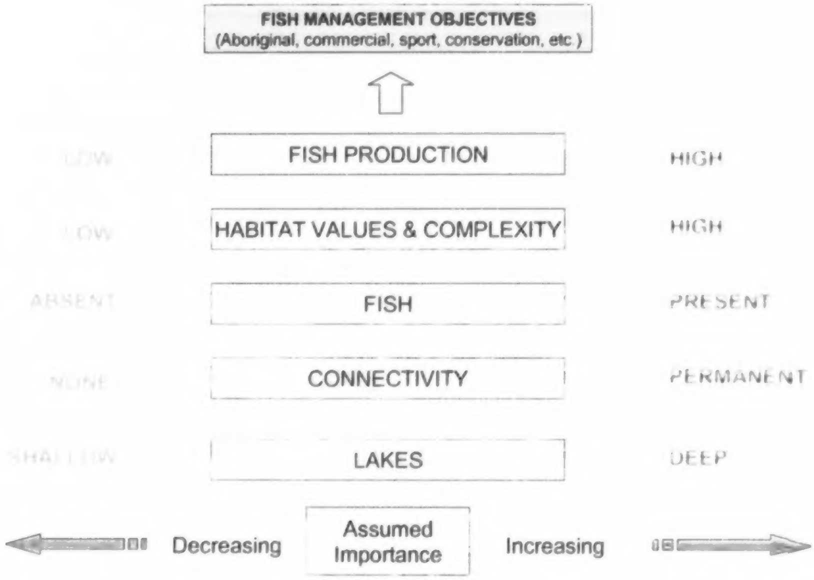
FISH MANAGEMENT OBJECTIVES
(Aboriginal, commercial, sport, conservation, etc.)
LOW
FISH PRODUCTION
HIGH
LOW
HABITAT VALUES & COMPLEXITY
HIGH
ABSENT
FISH
PRESENT
NONE
CONNECTIVITY
PERMANENT
SHALLOW
LAKES
DEEP
Decreasing
Assumed Importance
Increasing

Figure 29. Initial considerations of lake importance in relation to fish management objectives (to be assessed in relation to compensatory options)

The use of risk management (RMF) and "Pathways of Effects" models (Fisheries and Oceans Canada 2004*b*) would assist in the identification of activities that would impact upon fish habitat that may be mitigated and those which cannot. Progressive compensatory and restorative measures should be used so that environmental benefits may accrue during development activities rather than delaying them until completion of the development. Such measures would be chosen as appropriate, but consideration must be given to the end product to realize a no net loss in productive capacity of fish habitat. Accordingly, indices of production at different scales of biological organization and their linkages must be identified as criteria for success in meeting objectives. Thus the value of habitat would be progressively restored rather than deferred for protracted periods or indefinitely.

In relation to the Arctic and other regions where ecological information is scarce or lacking, the risk in accepting loss is compounded by this paucity of knowledge. Thus the

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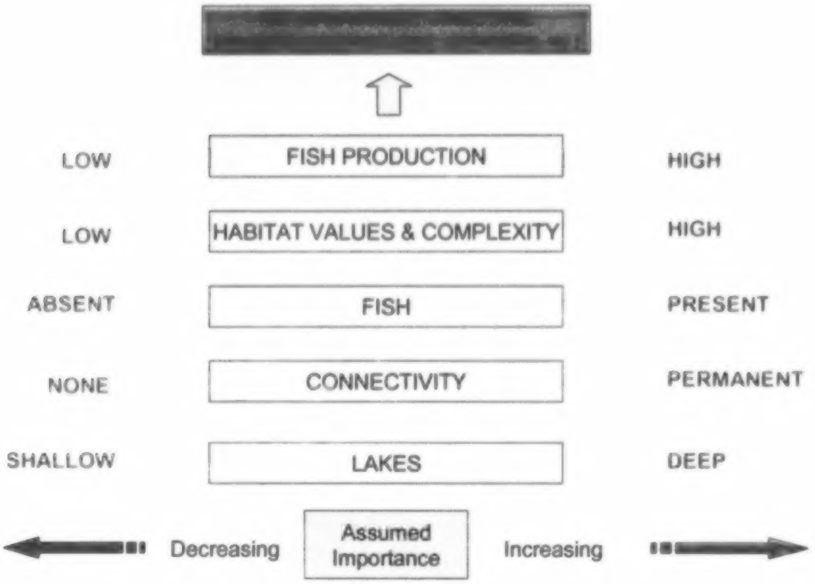

Figure 29. Initial considerations of lake importance in relation to fish management objectives (to be assessed in relation to compensatory options).

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In relation to the Arctic and other regions where ecological information is scarce or lacking, the risk in accepting loss is compounded by this paucity of knowledge. Thus the decision is one including much uncertainty and of high risk and as such, it has inherent experimental value. Under these circumstances new information must be obtained. It is suggested that this should be over the short- and the long-term. It should be a joint responsibility of the industry and other stakeholders, and the generation of knowledge through research should occur via a consortium of these parties and possibly others.

Funding for this research and monitoring should be the primary responsibility of the proponent of the development but collaborative studies would be for mutual benefit of those within the consortium and the public. In this context, if habitat compensation cannot be accomplished or the damage is deemed irreparable (i.e. cannot be restored or compensated for in the area of the development), and options are seriously limited, accepting funds as partial compensation could be considered. Such funds would be administered under the auspices of a consortium of stakeholders and could be directed towards the generation of new knowledge. While the acceptance of funds as partial compensation for damage to fish habitat is a departure from recognized choices its value lies in the opportunity to fund the obtaining of new knowledge that will benefit future decision making (McDaniels and Gregory 2004), and therefore reduce the risks that were present in the initial "experimental" decision (Minns et al. 1996).

Research will be required together with monitoring to assess the success of compensatory and restorative activities. This should be a component of authorizations where large-scale developments will impact fish habitat and/or where information is inadequate to predict the outcome of habitat change. Options to consider would include increasing the accessibility of habitats to fish, opportunities to access fish-less lakes, and changing physical features of lakes to add structural complexity and refuges, while striving to maintain the function of habitat (refer to Figure 30).

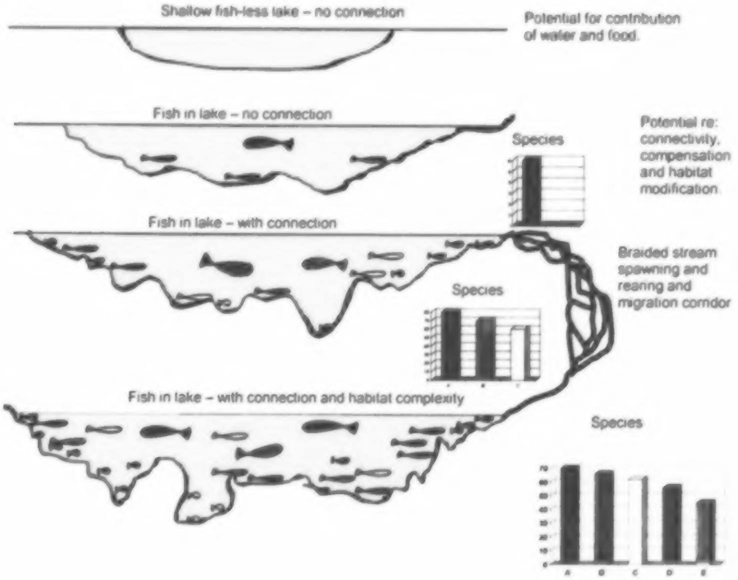
Shallow fish-less lake – no connection
Potential for contribution of water and food.
Fish in lake – no connection
Species
Potential re: connectivity, compensation and habitat modification.
Fish in lake – with connection
Braided stream spawning and rearing and migration corridor
Species
Fish in lake – with connection and habitat complexity
Species
70
60
50
40
30
20
10
0
A
B
C
D
E

Figure 30. Species diversity, habitat complexity, and linkages, for consideration of compensatory and restorative options.

The availability of relevant knowledge should drive the need for research and monitoring to reduce risks in decision making. As such, if decisions are made that incorporate the elimination of a lake the provision of scientifically-defensible knowledge should be required within the context of assessing the effects of this environmental change and the success of compensatory measures. In addition, if restorative actions are required research and monitoring should be undertaken in order that a flexible plan may be assessed through the meeting of objectives over the course of industrial activity. In both the restorative and compensatory aspects it is essential that evaluations of success (and failures) be reported along the path towards meeting regional and Departmental objectives.

CONCLUSIONS AND RECOMMENDATIONS

This section contains statements and recommendations deduced from the assessment of literature and from consultations. They are organized into the same broad categories used in the report, i.e. habitat science; industries; aspects of fish habitat management (including mitigation, compensation, and restoration); and policy and management. The categories are not necessarily independent of each other, for example the need for scientific research is a fundamental component of each.

CONCLUSIONS AND RECOMMENDATIONS OF PARAMOUNT IMPORTANCE

- **Damage to fish habitat in pristine regions of the Canadian Arctic is occurring and escalating and there is currently no assurance that habitat compensatory and restorative measures will be effective in meeting the Habitat Policy requirements regarding fish habitat productive capacity, which are linked to the *Fisheries Act* in order to sustain fish productivity.**

- **The inadequacy of even basic ecological knowledge, the absence of validations of habitat compensatory and restorative measures regarding habitat linkages to fish productivity, and no examples of whole lake restoration and compensation to guide developments forecast irreparable harm.**

- **Significantly more knowledge and understanding of the basic biology and habitat requirements of Arctic species are required, especially in view of the increasing human development in northern Canada, global environmental changes and the need to determine and assess their cumulative effects. Provision of this knowledge and understanding will assist decision making, reduce risk and uncertainty and facilitate mitigation, compensatory and restorative measures designed to sustain aquatic resources in this and other remote regions in Canada.**

- **Co-operative and collaborative ventures with industries, governments, academia and Aboriginal communities should provide for the acquisition of new**

information. Opportunities currently exist for this to occur. Multi-stakeholder consortia should be the mechanisms for fund acquisition and disbursement.

- **The successful application of the Habitat Policy to developments in regions of Canada where there is a paucity or absence of basic information for making sound decisions is jeopardized in view of the potential uncertainty of success of habitat mitigation, compensatory and restorative measures. In this context consideration must be given to learning from, and adapting to, experimental decision making. This will be accomplished through committed research and monitoring which is appropriately scaled over time and in space to meet stated Departmental objectives.**

- **Compensatory, restorative, and related techniques, measurements and evaluation, are priority research topics that meet with approval from all industry sectors, the scientific community and habitat practitioners.**

SCIENCE

Key conclusions

- **The Arctic represents unique habitat for fish because of its extreme biogeochemical and climatic conditions (extreme cold and ice cover of lakes and rivers, extreme variations in daylight, low nutrient supply and productivity, and low biodiversity).**

- **Arctic freshwater research has not received enough emphasis over the last 3 decades and as a result current development impacts on aquatic organisms cannot be easily, readily or accurately assessed.**

- **Information on the basic biology of Arctic fish, their dependence on, and interaction with, habitat is deficient, and as such, generally inadequate to meet the needs of habitat management.**

- **A number of human activities and human-influenced events (e.g. infrastructure development, mining, hydroelectric generation, oil and gas exploration and exploitation of renewable natural resources, seepage of toxic materials from abandoned sites, long-range transport of and deposition of atmospheric pollutants, and global change such as climate warming and ozone thinning) are affecting the Canadian Arctic environment.**

Key recommendations

- **Increase and commit to long-term, core, freshwater Arctic research programs related to fisheries-habitat science by:**

    - **Undertaking interdisciplinary studies to address the deficiencies in knowledge of fish-habitat interactions in the Arctic and other remote**

areas where habitat is being destroyed and altered through human development (use the pristine lakes to obtain basic ecological information as reference sites);

- **Undertaking ecosystem and sub-component research. Specific topics requiring attention include seasonal habitat refuges, critical habitat needs, habitat-fish interactions, fish-habitat modeling, habitat productivity surrogates, lake "fish-out" protocols and data manipulation for predictive purposes regarding yield and assemblages, controlling factors concerning survival and recruitment, specific habitat requirements of fish at different life stages, behavioral and physiological aspects linked to habitat use and optimal conditions, survival and function, population research in relation to habitat manipulation, experimental compensatory and restorative techniques and their validation, genetic diversity and adaptability of populations to habitat change such as ionic increases under oligotrophic circumstances;**

- **Initiating large-scale, long-term experiments using mining developments and pristine areas to determine fundamental fish-habitat interactions and improve predictive and decision-making capabilities for fish and habitat management (e.g. model approach on the Experimental Lakes initiative);**

- **Seeking modification to funding strategies by e.g. NSERC to provide more flexibility in funding all sources of scientific endeavors and include government agencies; and**

- **Establishing a consortium of stakeholders that would provide funds, evaluate, allocate and oversee priority research projects. (e.g. industry, Aboriginals, Indian and Northern Affairs Canada (INAC), Natural Resources Canada (NRCan), Environment Canada (EC), DFO, territorial governments).**

INDUSTRIES

Key conclusions

- **The three industrial mining sectors examined and metal mining have impacts on fish habitat which include the alteration and degradation of lake and stream habitat.**

- **Alteration and degradation of habitat will continue. Quantifiable portions of fish habitat are impacted by the respective sectors.**

- **All the mining sectors examined are viable, and exploration and expansion of operations is continuing.**

- **There is every expectation that placer gold, oil sands, and diamond mining will continue to operate into the foreseeable future (>20 years). Placer, oil sands, and diamond mining have been in existence for over 100, 60, and 10 years, respectively.**

- **Differences of opinion exist among some industry sectors and regulators regarding their impact on fish habitat (issues that could be alleviated through unbiased independent reviews).**

- **All three industrial sectors expressed a positive interest in, and would welcome participation in, mutually-beneficial research projects, especially those designed around the issues of compensation and restoration.**

ASPECTS OF FISH HABITAT MANAGEMENT (INCLUDING MITIGATION, COMPENSATION, AND RESTORATION)

Key conclusions

- **The success of mitigation, compensatory and restorative measures is intimately linked to knowledge of the basic needs and uses of habitats by fish. Aside from the need for rigorous evaluations of the efficacy of compensatory measures, the absence or paucity of this information in the Arctic, and in other remote locations, is one of the fundamental problems facing successful mitigation, compensatory, and restorative efforts.**

- **Recent examinations of the efficacy of fish habitat compensatory measures and their evaluations on a national level revealed inadequate assessments and compliance resulting in an overall loss of fish habitat. Lack of scientific rigor, inadequacy of knowledge, and inadequate assessments were constraints to realizing Departmental objectives.**

- **There has been no aquatic habitat "restoration" of a diamond mine pit, and very limited and rudimentary restoration of placer gold mining areas.**

- **Compensatory and restorative measures and monitoring regarding fish habitat are required from diamond, oil sands, and placer mining sectors in relation to their activities. These activities should occur throughout the duration of mining and, wherever possible, not be deferred until it ceases.**

- **There have been no complete evaluations of habitat compensatory initiatives at diamond mining sites. They have yet to be proved effective and successful.**

- **Money (i.e. $1.5 million) has been received as compensation for some of the impacts due to diamond mining operations. In addition, compensatory measures have been implemented and are being monitored.**

- **The Habitat Policy's "hierarchy of preferences" to compensate for lost habitat on a "like-for-like" basis is seriously limited in areas where information deficiencies exist and there is uncertainty of success. "Like-for-like" habitat compensation is not considered to be a viable option when the likelihood of success is low.**

- **Significant money and effort is being expended by certain industries (i.e. diamond and oil sands mining) to monitor the changes to aquatic conditions creating large databases. Their value will be apparent through time and therefore analysis should be an ongoing and adaptive requirement.**

- **There has been no field evaluation and development of methods used to quantify fish habitat in the Arctic in relation to diamond mining.**

- **The use of an array of surrogates for habitat productive capacity and models used to link habitat and fish use and production require validation in the Arctic.**

- **Compensatory options are often limited by knowledge and constrained by physical features of the land. There has been no recreation of a lake as compensation for one that has been eliminated, but creation of a lake as compensation for the elimination of streams is proposed for oil sands developments.**

- **Successful compensation for lost and degraded habitat in the mining sectors examined is considered to have occurred in a few circumstances.**

Key recommendations

- **Scientific rigor is required in the design, and the evaluation of habitat compensatory and restorative projects.**

- **Scientific standard methodologies should be used in assessing fish yield and communities in lakes to be eliminated to build a database that will permit better predictions and understanding.**

- **Evaluation of projects that will significantly impact aquatic habitats must combine monitoring and research components in a defensible and flexible (adaptive) manner, over time frames sufficient to meet management and scientific needs. A combination of short- and long-term evaluations is required.**

- **Rehabilitation (which improves a system to a good working order) should be considered as a more practical, initial objective than the restoration of fish habitat (which returns an ecosystem to a close approximation of its condition before it was disturbed), both of which require scientific knowledge and understanding.**

- **Objectives that are linked to meet fish and habitat management plans must be described so that success may be evaluated along a path(s) towards them, and adapted as required.**

- **Evaluation of habitat compensatory and restorative measures should encompass different levels of biological organization and trophic status, while recognizing the importance of population variability, threshold responses, and the accuracy of the metrics employed. A national data management system with easy access for data retrieval and assessment is required for such activities and research information. At the very least this should be for the boreal-Arctic regions.**

- **Climate change issues must be recognized and addressed in the design of monitoring and research programs for areas that are especially vulnerable to such change (e.g. Arctic systems).**

- **Restoration activities should, wherever feasible, be undertaken during the operation of mining and not await its termination (e.g. the sequencing and decommissioning of open pits and their restoration).**

POLICY AND MANAGEMENT

Key conclusions

- **There has been inconsistent application of the *Fisheries Act* among the mining sectors.**

- **Placer mining has been permitted to affect fish habitat under a section 35 "class authorization," oil sands and diamond mining are permitted to operate under site-specific authorizations, lakes used in metal mining as tailings impoundment areas (TIAs) are regulated through the section 36 Metal Mining Effluent Regulations.**

- **Decisions to use lakes in metal mining for TIAs must be authorized by the Governor in Council (cabinet), whereas similar decisions regarding diamond mining are made by the Fisheries Minister or designate.**

- **There tends to be an absence or scarcity of fish management and habitat plans for areas where mining development is occurring in northern Canada.**

- **Information on, for example, freshwater Arctic research and management programs, appears fragmented and requires coordination at regional and national levels.**

- **There has been a lack of standardized approaches used by habitat practitioners to evaluate and audit alteration and degradation of habitat, and compensatory and restorative measures in Canada.**

- **Guidelines have been produced that assist habitat practitioners in making decisions regarding developments that will impact habitat.**

Key recommendations

- **Management decisions, and especially those relating to major developments that will significantly impact fish habitat, and that include much inherent uncertainty and therefore a high level of risk, must be viewed as experimental. They should be amenable to evaluation and be adaptable to derive better knowledge and thereby help to guide future decisions.**

- **Fundamental habitat area and fish management plans and objectives are required that will guide decisions regarding the destruction of fish habitat, compensatory and restorative measures.**

- **Clear objectives are required that will permit both management decisions and operational activities to be evaluated over time frames sufficient for their full assessment. The goals of the Habitat Policy and the needs to meet the mandate of the Department lie within this context. Conservation of fish habitat and fish is one such goal.**

- **A number of new and revised "guidance documents" is required regarding the use of selected metrics of habitat productivity, compensatory and restorative measures and evaluation criteria.**

- **The guide relating to the application of the Habitat Policy and the issues around NNL should be updated to incorporate new development circumstances (e.g. in northern and other remote areas in Canada where knowledge and understanding of fish and aquatic habitat is limited).**

- **Authorizations for major industrial developments in regions where knowledge of impacts, their mitigation, compensation and restoration is low, must include as part of the proposal the provision for long-term monitoring, research and assessment that may include experimental fish habitat manipulations to generate new knowledge of importance to future decision making.**

- **If a development is agreed upon by affected parties and allowed to proceed it is important that this acceptance includes all aspects of mitigation, compensatory and restorative actions.**

- **Financial compensation that is linked to fish and habitat area plans should be an option for consideration only in those circumstances where the requisite information base is low and habitat compensation likely to be uncertain or impossible. Risk evaluations must be part of this scenario. Funding, in part, could be independently managed and directed for research and monitoring of the development, and habitat compensatory and restorative projects.**

- **The transfer of scientific information regarding techniques and methods to assist habitat practitioners must continue through meetings and information transfer to promote consistency of approaches to habitat management at national and regional scales.**

- **Habitat practitioners' workshops are required to assist in the application of scientific methodologies on a regular basis and, similarly, to evaluate performance in relation to decisions to achieve the goals of the Habitat Policy.**

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We are indebted to the many people who voluntarily provided valuable and insightful comments for use in this report. The diversity of their interests and knowledge together with the published scientific findings provided us with a measure of comfort regarding the significance of the deductions from this review. We apologize to these people for not being able to use all of their comments for, like the scientific literature we examined, the enormity of the database precluded the inclusion of all information (the complete set of comments is contained within the report by Samis et al. (2005)). We are hopeful however, that we captured the significant points from the information we received and obtained.

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